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OZONE POLLUTION: Impacts on carbon sequestration in Europe



Report prepared by
the ICP Vegetation
April, 2012

Editors:
Harry Harmens
and
Gina Mills

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Working Group on Effects
of the
Convention on Long-range Transboundary Air Pollution

Ozone Pollution: Impacts on carbon sequestration in Europe

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Harry Harmens and Gina Mills (Editors)

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Executive summary

Since the industrial revolution, concentrations of carbon dioxide (CO₂) in the atmosphere have been rising, initially slowly but in recent decades more rapidly. This is primarily due to an increase in fossil fuel burning associated with population growth and enhanced social and economic development. In recent decades deforestation, especially in the tropics, has also contributed considerably to the rise in atmospheric CO₂ as tropical forests are a major sink for CO₂. The rise in atmospheric CO₂ concentrations has resulted in a rise in the surface temperature of the earth (global warming). In addition to CO₂ increasing, the atmospheric concentrations of other gases contributing to global warming (greenhouse gases) such as nitrous oxide, methane, halocarbons and ozone have risen too. Depending on future scenarios, the earth's surface temperature is predicted to rise by a further ca. 2 – 4°C by the end of the 21st century. Currently, ozone is considered to be the third most important greenhouse gas, after CO₂ and methane. In contrast to CO₂ and halocarbons, ozone is a short-lived greenhouse gas, so any reductions in ground-level ozone production will reduce atmospheric ozone concentrations within months and hence reduce its contribution to global warming. Long-lived greenhouse gases will stay in the atmosphere for a long time, so even when emissions are kept constant at the 2000 level, a further rise in surface temperature of 0.5°C is predicted by the end of the 21st century. In this study, we have investigated how ozone pollution is currently, and likely in the future to continue to be, reducing carbon (C) sequestration in the living biomass of trees (and other vegetation), thereby potentially exacerbating global warming.

Ozone pollution

As well as being a greenhouse gas, ozone is also an important atmospheric pollutant and has adverse effects on human health and the environment. Ozone is a naturally occurring chemical that can be found in both the stratosphere (as the so-called "ozone layer", 10 - 40 km above the Earth) and the troposphere (at "ground level", 0 - 10 km above the Earth). At ground level there is always a background concentration of ozone resulting from natural sources of the precursors and stratospheric incursions. Of concern for human health and vegetation (including C sequestration and food production) is the additional tropospheric ozone which is formed from complex photochemical reactions from fossil fuel burning in industrial and transport activities. As a result of these emissions, there has been a steady rise in the background ozone concentration in Europe since the 1950s to the current 30 – 40 ppb. Background ozone concentrations in Europe are still rising and predicted to rise until at least 2030, in part due to hemispheric transport of the precursors of ozone from other parts (developed and developing areas) of the world. Background concentrations in Europe have now reached levels where they have adverse impacts on vegetation. During periods of hot dry weather and stable air pressure, ozone episodes occur where concentrations rise above 60 ppb for several days at a time.

Vegetation as a sink for atmospheric CO₂ and ozone

Atmospheric gases such as CO₂, ozone and water vapour are exchanged through microscopic stomatal pores on leaves. This for instance enables plants to fix CO₂ for photosynthesis and hence growth, and to transpire for the adjustment of the internal water balance. The more open the stomata are, the more CO₂ and ozone will enter the plant and the more water will transpire. Ozone entering the plant has the potential to damage plant cells by forming reactive oxygen species, which can lead to detrimental effects on photosynthesis and growth and/or ultimately to cell death. The magnitude of these damaging effects depends on the plant species and genotype, concentration of ozone, duration of exposure, climate and soil conditions. Plants are able to detoxify a certain amount of ozone, but above this amount damage to vegetation is likely to occur, either as acute damage due to exposure to

'high' ozone concentrations that usually occur during ozone episodes or as chronic damage due to prolonged exposure to elevated background ozone concentrations. Hence, terrestrial vegetation is considered an important sink for the greenhouse gases CO₂ and ozone. However, if ozone concentrations are high enough to reduce photosynthesis (i.e. CO₂ fixation) and/or above-ground plant growth, then less CO₂ and ozone will be taken up by the vegetation, leading to a positive feedback to atmospheric CO₂ and ozone concentrations and therefore global warming.

Ozone impacts in a changing climate

The future impacts of ozone on C sequestration in European terrestrial ecosystems will depend on the interaction with and magnitude of the change of the physical and pollution climate, represented by rising temperatures, increased drought frequency, enhanced atmospheric CO₂ concentration and reduced nitrogen deposition. Ecosystems are inherently complex, and for any one aspect of functioning, there are multitudes of driving factors. Exposure studies on the interaction between ozone and other pollutants (nitrogen) and climate change often show the following:

- **Elevated CO₂ concentrations** – Elevated ozone and CO₂ often affect plant physiology and soil processes in opposite directions. Hence, the overall response and resulting impact on C sequestration might well be cancelled out when both gases are enriched in the atmosphere.
- **Warming** – A rise in temperature stimulates ozone formation and directly affects the stomatal uptake of ozone since this process is temperature dependent. Warming can also indirectly affect the uptake of ozone via impacts on relative humidity, plant development and soil water availability, all of which influence the stomatal gas exchange. Some studies have shown that atmospheric ozone concentrations modify the response of plant species and genotypes to warming.
- **Enhanced drought** – It has often been postulated that drought will protect vegetation from ozone damage as the stomatal pores shut down more during periods of drought to prevent water loss. However, the interactions between ozone and drought (mediated via plant hormones) are more complex than first thought and drought might not protect ozone sensitive species from adverse impacts of ozone.
- **Nitrogen deposition** – Relatively few studies have investigated the impacts of both ozone and nitrogen on vegetation. Evidence suggests that ozone and nitrogen can have both synergistic and antagonistic effects on species and ecosystem processes, and that they may interact in unpredictable ways to affect plant communities.

Relatively few studies have investigated the interactive impacts of two or more drivers of change. The outcome of such studies often indicates complex interactions and non-linearity in responses. There is an urgent need for more field-based, larger scale experiments where vegetation is exposed to multiple drivers of climate change for several years (at least one decade) to further investigate the overall impact of a combination of drivers of change on terrestrial ecosystems. Modelling studies to predict future impacts of change should also be based on a multifactorial approach. So far, the impacts of ozone on vegetation and feedbacks to the climate have hardly been considered in global climate modelling. Recent modelling studies have shown that the indirect impact of ozone on global warming via its impacts on vegetation might be contributing as much to global warming as its direct effect as a greenhouse gas.

Impacts of ozone on C sequestration in the living biomass of trees

First flux-based assessment for Europe for the current (2000) and future climate (2040)

The DO₃SE (Deposition of Ozone for Stomatal Exchange) model was applied to estimate the magnitude of the impact of ambient ozone (in comparison with pre-industrial ozone concentrations) on C storage in the living biomass of trees. The Phytotoxic Ozone Dose above a threshold value of Y nmol m⁻² s⁻¹ (POD_Y) was calculated applying known flux-effect relationships for various tree species.

The following input data were used:

- i) Ozone and meteorological data provided by EMEP for the year 2000, and ii) ozone and climate data provided by the Rossby Centre regional Atmospheric climate model (RCA3) for current (2000) and future (2040) years.
- Land cover data to identify the distribution of forest tree species: i) for EMEP data the species-specific JRC land cover data and for ii) RCA data the UNECE Long-Range Transboundary Air Pollution (LRTAP) Convention harmonised land cover data were used.
- Forest C stock data were derived from the European forests inventory dataset.

In addition, for the year 2000 using EMEP ozone and meteorological data, the application of generic parameterisations for trees in DO₃SE (POD₁) were compared with the application of climate region specific parameterisations (a mixture of POD₁ and POD_{1.6}) and a deactivated soil moisture deficit (SMD) module (POD₁), i.e. no limitation of soil moisture on stomatal conductance and hence ozone flux (no influence of drought). Finally, the flux-based results were compared with concentration-based results (AOT40²) to highlight the differences.

Table 1 Estimated total reduction of C storage (Mt C) in the living biomass of trees due to ozone in 2000 and 2040 compared to pre-industrial atmospheric ozone levels. Values in brackets show the percentage of estimated reduction. Key: Cl. Specific = climate specific, SMDoff = soil moisture deficit module switched off (see text for further details), NE = Northern Europe, ACE = Atlantic Central Europe, CCE = Continental Central Europe, ME = Mediterranean Europe.

Input data	Year	DO ₃ SE parameterisation	POD _Y or AOT40	NE	ACE	CCE	ME	Total
EMEP	2000	Generic	POD ₁	256 (8.5)	40.3 (14.3)	735 (14.1)	218 (11.9)	1249 (12.0)
	2000	Cl. specific	POD _{1/1.6}	553 (14.5)	44.3 (15.0)	877 (14.8)	215 (10.7)	1689 (13.7)
	2000	SMD _{off}	POD ₁	268 (8.8)	43.1 (15.1)	1034 (19.7)	437 (23.4)	1782 (17.3)
	2000	Generic	AOT40	29.7 (1.3)	10.5 (2.8)	499 (10.3)	253 (14.2)	792 (8.2)
RCA	2000	Generic	POD ₁	317 (10.4)	60.6 (21.2)	1247 (21.4)	305 (14.7)	1929 (16.2)
	2000	Generic	AOT40	41.8 (2.1)	13.2 (5.0)	483 (10.4)	254 (14.1)	791 (8.4)
	2040	Generic	POD ₁	271 (9.3)	52.9 (19.8)	821 (15.7)	184 (10.2)	1330 (12.6)
	2040	Generic	AOT40	-0.9 (-0.3)	4.4 (2.3)	68.5 (2.1)	57.1 (4.1)	129 (2.1)

The main results are (Table 1):

- When applying the flux-based methodology and a generic parameterisation for deciduous and conifer trees, a reduction of C sequestration in the living biomass of trees by 12.0 (EMEP input data) to 16.2% (RCA input data) was calculated in comparison with pre-industrial ozone impacts. The flux-based approach indicates a high risk of ozone impacts on forests in Atlantic

² The accumulated hourly mean ozone concentration above 40 ppb, during daylight hours

and Continental Central Europe, and also a considerable risk in northern Europe (in comparison with the concentration based approach).

- The climate-region specific parameterisation for 2000 revealed higher C reductions (13.7%) due to ozone compared to the generic parameterisation (12.0%) for calculating POD_Y .
- The deactivation of the soil moisture deficit module of the DO_3SE model, which simulates drought-free stomatal ozone uptake conditions throughout Europe, led to an increase in C reduction, especially in the warmer and drier climates in Central and Mediterranean Europe.
- In the future (by 2040) the reduction of C storage in the living biomass of trees due to ozone is expected to decrease considerably compared to the reduction in 2000, mainly as a result of a predicted reduction in atmospheric ozone concentrations across Europe. The decrease is smaller for the flux-based than the concentration-based approach.
- The concentration-based approach (AOT40) predicts substantially lower C reductions compared to the flux-based approach (POD_Y), especially in Northern and Continental Central Europe and when using RCA input data. In 2000, the percentage reduction (ca. 8%) in C storage in the living biomass of trees when applying the ozone concentration-based approach is similar to the percentage reduction (ca. 10%) calculated for forests in northern and central Europe in Chapter 6 and similar to values reported in the literature.

Whilst the spatial patterns and temporal trends indicated above can be postulated with a considerable degree of certainty, the absolute figures of C reductions given in this report have to be interpreted very carefully. It should be remembered that these are for effects on living tree biomass only, and do not take into account any effect on soil C processes, including any direct or indirect ozone effects on below-ground processes that affect the rate of C turnover in the soil. Furthermore, the response functions used were derived for young trees (up to 10 years of age). However, there is some scientific evidence from epidemiological studies that the functions are applicable to mature trees within forests.

Case study in northern and central Europe applying the AOT40 method

A more detailed study based on relative growth rates of trees was conducted to assess the impacts of current ambient atmospheric ozone concentration (in comparison to pre-industrial ozone levels in the range of 10 -15 ppb) on C sequestration in the living biomass of trees in temperate and boreal forests. Using data from forests inventories on forest types, age classes and structure, growth and harvest rates and combining these with AOT40-based dose response relationships for young trees, calculated yearly growth increment values were converted to C stock changes. The estimated percent reduction in the change of the living biomass C stock across forests in ten countries was 10%. However, for different countries these values ranged between 2 and 32% (**Table 2**).

Table 2 Estimated reductions in annual C sequestration due to current ambient ozone exposure as compared to pre-industrial ozone levels in northern and central Europe.

Country	Decline (%)	Country	Decline (%)	Country	Decline (%)
Czech Republic	32.0	Finland	2.2	Lithuania	13.8
Denmark	5.8	Germany	12.3	Norway	1.8
Estonia	4.5	Latvia	8.8	Poland	12.8
All countries	9.8			Sweden	8.6

The most important factor that determines the changes in the forest living biomass C stock is the gap between growth and harvest rates. If this gap is small, then a certain growth reduction caused by ozone will have a relatively large impact on the C stock change, and vice versa. By far the most important countries for C sequestration in the living biomass C stocks in northern and central Europe are Sweden, Finland, Poland and Germany. Ozone-induced growth reductions will also result in an economic loss for forest owners.

A global perspective of impacts on C storage in terrestrial ecosystems

The JULES (Joint UK Land Environment Simulator) model has been run with ozone fields and observed climatology over the period 1901-2040 to assess the impacts of ozone on the global C and water cycle. In JULES, the plant damage due to ozone directly reduces plant photosynthesis, and thereby indirectly, leaf stomatal conductance. With elevated near surface ozone levels, the model simulates decreased plant productivity, and as less CO₂ is required for photosynthesis, reduced stomatal conductance. Therefore, the plant is able to preserve water supplies. However, some recent studies have shown that ozone impairs stomatal functioning such that ozone might enhance rather than reduce stomatal conductance. As no direct effect of ozone on stomatal functioning is currently incorporated into JULES, the indirect effect of ozone on stomata via photosynthesis was switched off ('fixed stomata') in the current study to investigate the consequences for the global C and water cycle. In JULES, the ozone flux-based method was applied.

Table 3 Simulated future percentages changes (% Δ) in carbon (C) and water cycle (runoff) variables globally for three time periods: 1901-2040, 1901-2000 and 2000-2040. GPP = Gross Primary Productivity, Veg = vegetation, Gs = stomatal conductance (Scenario: SRES A2).

1901-2040	% Δ GPP	% Δ VegC	% Δ SoilC	% Δ TotalC	% Δ Runoff	% Δ Gs
Control	-15.4	-10.9	-9.7	-10.0	12.6	-13.3
Fixed stomata	-17.9	-11.8	-10.5	-10.9	1.4	-1.6
2000-2040						
Control	-6.9	-5.0	-4.1	-4.4	4.5	-5.0
Fixed stomata	-8.1	-5.5	-4.6	-4.8	0.6	-0.5
1901-2000						
Control	-9.2	-6.2	-5.8	-5.9	7.7	-8.7
Fixed stomata	-10.7	-6.7	-6.2	-6.4	0.8	-1.1

Applying ozone stomatal flux response relationships in JULES, the model predicted that the reduction in C stored in vegetation is 6.2% globally and almost 4% in Europe in 2000 compared to 1900, and is predicted to rise to 10.9% globally and ca. 5 to 6% in Europe by 2040 (**Table 3**) due to a predicted rise in atmospheric ozone concentrations in the future emission scenario applied. As expected, results from the control run suggest a large indirect effect of ozone (via photosynthesis) on stomatal conductance and runoff. Unsurprisingly, stomatal conductance and river runoff changed little through time in the fixed stomata simulation, where the indirect effect of ozone on stomata via photosynthesis was switched off. However, despite the difference in stomatal conductance response between simulations, the differences in the response of the C cycle are rather modest. It can be concluded that in the absence of a direct effect of ozone on stomatal conductance, ozone-vegetation impacts act to increase river runoff and freshwater availability substantially due to a reduced water loss from soil via transpiration from vegetation. However, such an increase might not occur if ozone has adverse impacts on stomatal functioning, reducing their responsiveness to environmental stimuli.

In addition, we analysed the impacts of ozone and drought interactions on plant productivity in Europe by applying the climate specifically for the year 2003, which was a very dry year across the whole of Europe. Large reductions in plant productivity were simulated under drought conditions. The net impact of ozone is to further reduce plant productivity under drought. In the absence of a direct effect of ozone on stomatal conductance, ozone acts to partially offset drought effects on vegetation.

The focus of this report has been on quantifying impacts of ozone pollution on C sequestration in the biomass of forest trees. The quantification of the dynamics of C sequestration within agricultural and grassland systems is complicated, with balances for crops often considered to be zero. In this report we have discussed how ozone has the potential to further complicate the C balance of crops by reducing CO₂ fixation, increasing repair respiration, and reduce biomass allocation to the harvestable crop, stubble and roots. There is also some evidence of ozone indirectly impacting on the microbial community within soils via effects on the crop.

Recommendations

Policy More stringent reductions of the emissions of precursors of ozone are required across the globe to further reduce both peak and background concentrations of ozone and hence reduce the threat from ozone pollution to C sequestration. It would be of benefit to better integrate policies and abatement measures aimed at reducing air pollution and climate change as both will affect C sequestration in the future. Improved quantification of impacts of ozone within the context of climate change is urgently required to facilitate improved future predictions of the impacts of ozone on C storage in the living biomass of trees. Stringent abatement policies aimed at short-lived climate forcers such as ozone provide an almost immediate benefit for their contribution to global warming.

Research There is an urgent need for more field-based, larger scale experiments where vegetation is exposed to multiple drivers of climate change for several years (at least one decade) to further investigate the overall impact of a combination of drivers of change on C sequestration in terrestrial ecosystems. Further development of the ozone flux-based method and establishment of robust flux-effect relationships are required for additional tree species, in particular for those species representing the Mediterranean areas. Field-based ozone experiments should also include the assessment of ozone impacts on below-ground processes and soil C content. Further epidemiological studies on mature forest stands are required for the validation of existing and new ozone flux-effect relationships. Experiments are needed on the interacting effects of climate change and ozone, including quantifying impacts of reduced soil moisture availability, rising temperature and incidences of heat stress, impacts of rising CO₂ concentrations and declining nitrogen deposition. Impacts of other drivers of change on existing flux-effect relationships should be investigated. Further development of climate region-specific parameterisations for flux models is needed to improve the accuracy of predictions. Existing flux models (e.g. DO₃SE) will have to be further developed to include more mechanistic approaches for the accurate prediction of combined effects of ozone, other pollutants and climate change, on various plant physiological processes and hence C sequestration.

There is an urgent need to further include ozone as a driver of change in global climate change modelling to quantify its impact (either directly or indirectly via impacts on vegetation) on global warming. Such modelling should further investigate the mechanisms of interactions between ozone and other drivers of global warming. Finally, there is a need to quantify the economic impacts of ozone on forest growth in order to establish the economic consequences for the wood industry. In this light and for enhanced C storage in the living biomass in the future, the ozone-sensitivity of tree species and varieties should be considered as a factor in future breeding and forests management programmes.

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1 Introduction

Harry Harmens, Gina Mills

1.1 Background

Since the industrial revolution concentrations of carbon dioxide (CO₂) in the atmosphere have been rising, initially slowly but in recent decades more rapidly (IPCC, 2007). This is primarily due to an increase in fossil fuel burning, associated with population growth and enhanced social and economic development. However, in recent decades deforestation particularly in the tropics has also contributed considerably to the rise in atmospheric CO₂ as tropical forests are a major sink for atmospheric CO₂. The rise in atmospheric CO₂ concentrations has resulted in a rise in earth surface temperature (global warming). In addition to CO₂, the atmospheric concentrations of other greenhouse gases (i.e. gases contributing to global warming) such as nitrous oxide (N₂O), methane, halocarbons and ozone has risen too. Atmospheric concentrations of greenhouse gases increase when emissions are larger than removal processes. Depending on future scenarios, the earth temperature is predicted to rise by a further ca. 2 – 4°C by the end of the 21st century (IPCC, 2007). Currently, ozone is considered to be the third most important greenhouse gas, after CO₂ and methane. In contrast to CO₂ and halocarbons, ozone and methane are short-lived greenhouse gases (IPCC, 2007).

Apart from being a greenhouse gas, ozone is also an atmospheric pollutant and has adverse effects on human health (WHO, 2008) and the environment (e.g. Ashmore, 2005; Royal Society, 2008; Mills et al., 2011a). Recently, the adverse impacts of ozone on food production and associated economic losses were highlighted (Van Dingenen et al., 2009; Averny et al., 2011a,b; Mills and Harmens, 2011). Ozone is a naturally occurring chemical that can be found in both the stratosphere (the so-called "ozone layer", 10 - 40 km above the Earth) and the troposphere (the "ground level layer", 0 - 10 km above the Earth). Within the troposphere at vegetation level, there is always a background concentration of ozone resulting from natural sources of the precursors such as oxides of nitrogen (NO_x) and non-methane volatile organic compounds (NMVOCs) released from, for example soil and vegetation, as well as incursions of ozone from the stratosphere which occurs under certain meteorological conditions. Of concern for human health and vegetation is the additional tropospheric ozone which is formed from complex photochemical reactions involving NO_x, carbon monoxide and NMVOCs released from fossil fuel burning in industrial and transport activities. As a result of these emissions, there has been a steady rise in the background ozone concentration in Europe since the 1950s to the current 30 – 40 ppb (Royal Society, 2008). Background ozone concentrations in Europe are still rising and predicted to rise until at least 2030 due in part to hemispheric transport of the precursors of ozone from developing areas of the world (Royal Society, 2008). Background concentrations in Europe have now reached levels where they have adverse impacts on vegetation.

Ozone concentrations are usually highest in rural areas that are downwind of major conurbations where there are few other pollutants to react with ozone to reduce the concentration. They are also usually highest in spring and summer when temperature and light conditions are more conducive for ozone formation. These spring and summer peaks coincide with the growing season of most vegetation, increasing their vulnerability to ozone impacts. Due to the implementation of ozone precursor emission abatement policies, peak level ozone concentrations are gradually declining in Europe, North America and Japan, but not in the economically developing areas of the world such as central and southern Asia.

1.2 The role of the ICP Vegetation

The ICP Vegetation comprises over 200 scientists representing 34 countries of Europe and the USA and has its Programme Coordination Centre at the Centre for Ecology and Hydrology, Bangor, UK. The programme conducts experiments, analyses vegetation samples, synthesises current knowledge and develops models on current and future impacts of ozone on vegetation. In addition, the programme monitors and assesses the spatial distribution and temporal trends of the deposition of heavy metal, nitrogen and persistent organic pollutants (POPs) to vegetation across Europe (Harmens et al., 2011). The outputs of the programme contribute to the development of internationally-agreed protocols on pollution control by the Long-range Transboundary Air Pollution (LRTAP) Convention by providing scientific evidence and risk assessment methodology. This report covers one aspect of the work of the ozone group within the ICP Vegetation and contains reviews and data synthesis from participants in Austria, Sweden, Switzerland and the UK. In addition to this report, last year the ICP Vegetation Programme Coordination Centre put together a similar report on 'Ozone pollution: a hidden threat to food security' (Mills and Harmens, 2011). Next year the ICP Vegetation is planning to publish a report on the impacts of ozone on biodiversity and other ecosystem services, together with a report on heavy metal contamination and nitrogen enrichment in mosses in Europe. Other contributions to the Convention include predictions of ozone impacts on crops, (semi-)natural vegetation and trees in Europe for inclusion in the negotiations related to the revision of the Gothenburg Protocol to abate effects of acidification, eutrophication and ground-level ozone, biomonitoring studies of ozone impacts using sensitive indicator species, and a review of the potential use of mosses as biomonitors of POPs. For further details of the work of the ICP Vegetation we refer to its web site: <http://icpvegetation.ceh.ac.uk>.

1.3 Vegetation as a sink for atmospheric CO₂ and ozone

Both CO₂ and ozone are absorbed into plants via the thousands of microscopic stomatal pores on leaves which normally open during daylight hours to enable CO₂ fixation in photosynthesis and transpiration of water. The more open the stomata are (i.e. high stomatal conductance), the more CO₂ and ozone will enter the plant and the more water will transpire. Whereas CO₂ is used by the plant to enable growth, ozone has the potential to damage plant cells by forming reactive oxygen species, leading to reductions in key processes such as photosynthesis, growth and/or ultimately cell death. The magnitude of these damaging effects depends on the plant species, concentration of ozone, duration of exposure, climate and soil conditions. Plants are able to detoxify low concentrations of ozone to a certain level, but above this level damage to vegetation is likely to occur, either as acute damage due to exposure to 'high' ozone concentrations that usually occur during ozone episodes or as chronic damage due to prolonged exposure to elevated background ozone concentrations with or without ozone episodes (Mills and Harmens, 2011).

Hence, terrestrial vegetation is an important route through which the concentration of the greenhouse gases CO₂ and ozone can be reduced. However, if ozone concentrations are high enough to reduce photosynthesis (i.e. CO₂ fixation) and/or above-ground plant growth, then less CO₂ and ozone will be absorbed by the leaves of vegetation, leading to a positive feedback to atmospheric CO₂ and ozone concentrations and therefore more global warming. In a recent global climate modelling study, Sitch et al. (2007) concluded that the global warming effect of ozone due to this positive feedback could be at least as high as the direct global warming effect of ozone. In addition, they showed that ozone will reduce the terrestrial C sink, with the magnitude depending on the sensitivity of vegetation to ozone.

Agriculture, forestry and other land uses account for approximately 30% of the total anthropogenic greenhouse gas emissions (IPCC, 2007). Of these, agriculture accounts for about 60% of N_2O and 50% of CH_4 emissions, whereas deforestation and land use change are responsible for most of the CO_2 emissions (IPCC, 2007). Land use, land cover and changes in them have an important impact on the global C cycle. For example, deforestation is responsible for about 12% of the world's anthropogenic greenhouse gas emissions, whereas another 6% stems from peat oxidation and fires on degraded peatland areas (Van der Werf et al., 2009). The combined effects of logging and forest re-growth on abandoned land are responsible for 10-25% of global human-induced emissions (Achard et al., 2002; Gullison et al., 2007). Annual emissions from deforestation in Indonesia and Brazil alone equal four-fifths of the annual reduction target of the Kyoto Protocol (Santilli et al., 2005). Within land cover type there is also the potential to adapt management towards higher C sequestration potential, e.g. in forests by enhancing the gap between forest growth and harvest rates (Chapter 6).

1.4 Carbon sequestration in soils

The CO_2 taken up by vegetation will be sequestered in the shorter or longer term in plant material or soils. Soils are the largest C reservoir of the terrestrial C cycle. Worldwide they contain three to four times more organic C (1500 Gt to 1 m, 2500 Gt to 2 m depth) than vegetation (610 Gt) and twice or three times as much C as the atmosphere (750 Gt; Batjes and Sombroek, 1997). C storage in soils is the balance between the input of dead plant material (leaf and root litter, decaying wood) and losses from decomposition and mineralization of organic matter (heterotrophic respiration). Under aerobic conditions, most of the C entering the soil returns to the atmosphere by autotrophic root respiration and heterotrophic respiration (together called soil respiration). The mineralization rate is a function of moisture levels and chemical environment with factors such as temperature, pH, redox potential, nitrogen level and the cation exchange capacity of the minerals in the soil affecting the mineralization rate of soil organic C (Li et al., 1994; Kätterer et al., 1998; Reichstein et al., 2005; Heimann and Reichstein, 2008). Under anaerobic conditions, resulting from constantly high water levels, part of the C entering the soil is not fully mineralized and accumulates as peat.

Based on a limited number of studies, Guo and Gifford (2002) postulated that soil C stocks decline after land use changes from pasture to plantation (-10%), native forest to plantation (-13%), native forest to crop (-42%), and pasture to crop (-59%). Soil C stocks increase after land use changes from native forest to pasture (+ 8%), crop to pasture (+ 19%), crop to plantation (+ 18%), and crop to secondary forest (+ 53%). Most land use on peat soils requires drainage and is associated with a continuous loss of soil C stock. Laganier et al. (2010) concluded that the main factors that contribute to restoring soil organic C stocks after afforestation are: previous land use, tree species planted, soil clay content, pre-planting disturbance and, to a lesser extent, climatic zone. Hence, changes in land use have a significant impact on the C sequestration capacity of soils and the potential to be directed towards higher soil C sequestration.

1.5 Quantifying the damaging effects of ozone on vegetation

Many plant species (crops, trees and (semi-)natural vegetation) are sensitive to ozone within the range experienced in Europe. Ozone effects have been quantified by growing plants in:

- Open-top chambers. Impacts of ambient ozone can be quantified comparing plants exposed to lower than ambient ozone (by removing ozone via air filtration) and plants exposed to ambient ozone concentrations. In addition, plant exposed to ambient ozone can be compared to plants exposed to higher than ambient ozone concentrations (by adding ozone to either unfiltered or filtered ambient air).

- Field-release systems. Impacts of elevated ozone concentrations can be quantified by comparing plants exposed to ambient air and plants exposed to ozone-enriched air in field fumigation studies.
- Controlled environment chambers or greenhouses. Detailed plant physiological studies are often conducted in controlled environments to unravel the mechanisms of ozone impacts on vegetation without the confounding factor of a fluctuating climate.

Under the auspice of the ICP Vegetation, critical levels have been derived for crops, trees and (semi-) natural vegetation (LRTAP Convention, 2010). The critical levels used for the LRTAP Convention's Gothenburg Protocol to abate the effects of acidification, eutrophication and ground-level ozone were based on AOT40. Ozone exposures below 40 ppb were believed to be being detoxified by the plant's natural defence mechanisms and thus were not contributing to the damaging effects of ozone. Scientific research has developed further in the last decade, and currently the accumulated ozone flux via the stomatal pores on the leaf surface is considered to provide a more biologically sound method for describing observed effects. This new parameter is the Phytotoxic Ozone Dose above a threshold of Y , POD_Y (previously described as $AF_{st}Y$). It is calculated from modelling the effects of climate (temperature, humidity, light), ozone, soil (moisture availability) and plant development (growth stage) on the extent of opening of the stomatal pores, and like AOT40 is accumulated over a threshold, in this case a flux of $Y \text{ nmol m}^{-2} \text{ s}^{-1}$. Recently, the flux-based critical levels for vegetation were revised (LRTAP Convention, 2010; Mills et al., 2011a). It should be noted that the critical levels and response functions for tree species have been mainly derived from data from central and northern Europe. The risk of adverse ozone impacts on vegetation in the Mediterranean area is more uncertain as Mediterranean data are still scarce and flux models and/or flux-effect relationships are still being developed for Mediterranean vegetation. We refer to Chapter 4 (Table 4.5) for further details on trees species for which ozone flux-effect relationships were applied in this study.

1.6 Aims and content of this report

Aims:

- To review current knowledge on the potential impacts of ozone on C sequestration and ozone absorption by vegetation (particularly trees) and implications for climate change.
- To estimate the impacts of ozone on C storage in the 'living' biomass (including wood) of forests in Europe using flux-based and concentration-based methods.
- To model the impacts of ozone on the global C and water cycle.
- To discuss the potential impacts of ozone on croplands and grasslands.

Firstly, we summarise current knowledge of the deposition of ozone to vegetation and its potential effects on C sequestration via impacts on plant physiological processes (Chapter 2). In Chapter 3 we summarise the knowledge of ozone effects in a changing climate. An empirical flux-effect modelling approach (DO_3SE) was applied in Chapter 4 to provide maps showing the spatial distribution of both ozone (year 2000) and ozone & climate change (year 2040 with year 2000 as reference year) impacts on C storage in the biomass of forests in the Europe. In Chapter 5 the effects of ozone on C sequestration were further investigated by applying a land-surface model (JULES) and estimating the consequences for the global C and hydrological cycles. The current impacts of ambient ozone concentrations on C sequestration in forests were investigated in more detail for northern and central Europe (Chapter 6). Potential impacts of ozone on C sequestration in croplands and grasslands are described Chapter 7. In the final chapter, we summarise the impacts of ozone on C sequestration and consider the policy implications of the findings of this study.

2 Deposition of ozone to vegetation and its potential effects on carbon sequestration

Neil Cape, Sally Wilkinson, Harry Harmens, Gina Mills

2.1 Vegetation as a sink for ozone: non-stomatal and stomatal deposition

Ozone is removed from the atmosphere at the earth surface and vegetation is an important sink for ozone. The rate of removal depends on supply of ozone to the surface (atmospheric transport) and the effective surface area for removal to occur – the effective surface area may be much greater than a measured leaf area index (LAI). The loss of ozone above a vegetated surface can be attributed to the stomatal (i.e. via thousands of microscopic pores (stomata) on the leaf surface) and/or non-stomatal pathway. If the rate of ozone removal is the same as that for stomatal water vapour emission (after correction for the different gaseous diffusion rates of ozone and water vapour) then the whole flux is designated as 'stomatal'. In practice, at the canopy level ozone fluxes generally exceed the theoretical stomatal flux, and the difference is designated 'non-stomatal'.

Non-reactive removal of ozone occurs at rates proportional to the effective surface area for a wide range of different surface composition (Cape et al., 2009). If bare soil is present, deposition will also occur, at rates dependent on the accessible surface area (Gusten et al., 1996; Stella et al., 2011). Removal rates vary throughout the canopy depending on turbulence (air penetration) and the physiological status of leaves at different levels in the canopy. In addition to removal at surfaces, removal of ozone may also occur by chemical reaction with gases emitted by soils or plants. Reaction with nitric oxide (NO) from soils may enhance removal rates at the soil surface, but is unlikely to contribute greatly to removal below a plant canopy. However, emissions of reactive, biogenic volatile organic compounds (BVOCs) from plants may account for a significant fraction of ozone removal by a plant canopy before the ozone reaches the plant or soil surface (Goldstein et al., 2004). A recent study showed that ground-level ozone concentrations are influenced by circadian control of isoprene emissions from ecosystems (Hewitt et al., 2011). Ozone can cause subtle changes in leaf surface structures (Guenthard-Goerg and Keller, 1987) and might affect the structure of waxes on leaf surfaces (Mankovska et al., 2005).

Measurements show that the fraction of non-stomatal ozone loss varies with vegetation type, time of year, and weather conditions. The most obvious factors affecting the overall loss rates are the structure of the vegetation (tall, short; open/closed canopy) and the leaf area, which for most crop plants and deciduous trees has a marked seasonal variation. Although at the height of the growing season for a crop species most of the ozone uptake may be through stomata, at other times of the year, or in drier climates, stomatal ozone uptake may be much less than half of the total flux (Cieslik, 2004; **Table 2.1**). In actively growing canopies most of the ozone flux is to stomata unless the canopy is sufficiently open to allow transfer of ozone to understorey vegetation, or unless there are specific chemical sinks for ozone below the canopy, such as BVOCs. Modelling studies have shown that non-stomatal removal of ozone is not sufficient to reduce the flux of ozone through stomata, and so cannot serve as a protective mechanism for reducing leaf uptake of ozone (Altimir et al., 2008). A model (see Chapter 4) has been developed to estimate the stomatal uptake of ozone, depending on atmospheric ozone concentration, climatic conditions (light intensity, humidity, temperature), soil conditions (soil water potential or plant available water content) and plant developmental stage (Emberson et al., 2000a; LRTAP Convention, 2010).

Table 2.1 Examples of field measurements of ozone deposition in which stomatal and non-stomatal fluxes have been calculated.

Vegetation type	Conditions	Non-stomatal flux (% of total ozone flux)	Reference
Scots pine, Finland	Average over year	67%	Altimir et al. (2006)
Maritime pine, France	Summer daytime	20% (below canopy, higher with dew and at night)	Lamaud et al. (2002)
Norway spruce, Denmark	5-year period	> 90% in winter ca. 30% in summer	Mikkelsen et al. (2004)
Various, S. Europe	Mostly summer	31-88%	Cieslik (2004)
Subalpine forest, Colorado, USA	Summer daytime	20% (higher when wet)	Turnipseed et al. (2009)
Moorland, Scotland	Average over 4 years	70%	Fowler et al. (2001)
Intensive Grass, Germany	Summer, before and after cutting	20% before cutting 50% after cutting	Meszaros et al. (2009)
Potato, Scotland	Summer, daily median	10-80%	Coyle et al. (2009)
Wheat, Italy	May-June, anthesis to harvest	40-50% (more during senescence)	Gerosa et al. (2003)
Cotton, California	Summer	10-30% (higher when dry)	Grantz et al. (1997)
Grape, California	Summer	30-75% (higher when wet)	Grantz et al. (1995)

2.2 Role of BVOCs in the deposition, detoxification and impacts of ozone

Biogenic VOCs have been implicated in the removal of ozone within and close to plant canopies, as described above, but VOCs, including biogenic VOCs, also contribute to ozone formation on regional scales. The VOC isoprene is produced by many plant species, and provides protection against biotic and abiotic stresses (Laothawornkitkul, 2009). Globally, isoprene emissions from plants are estimated to far exceed anthropogenic emissions of VOCs (Guenther et al., 2006). Ozone production requires three components: nitrogen dioxide (NO_2), sunlight and a VOC. In sunlight, NO_2 molecules split into nitric oxide (NO) and an oxygen atom, which combines with molecular oxygen (O_2) to give ozone (O_3). The ozone formed, however, is short-lived because it will react with the NO to regenerate NO_2 and O_2 . In VOC-free air this cycle of reaction is in equilibrium, however, in the presence of UV light VOCs upset the 'simple' $\text{NO}/\text{NO}_2/\text{O}_3$ series of reactions by regenerating NO_2 from NO without consuming an ozone molecule, allowing ozone concentrations to increase. This process of linked cyclical reactions is known as 'photochemical ozone production' in the atmosphere. If ozone, or any other factor, leads to an increase in biogenic VOC emissions, then there is the possibility of a 'positive feedback' process. The evidence that ozone affects plants in a way that enhances VOC emissions such as isoprene and monoterpenes is limited and sometimes conflicting (Heiden et al., 1999; Rinnan et al., 2005; Tiiva et al., 2007). Large field-scale exposure of aspen trees to enhanced ozone showed increased isoprene emissions (Calfapietra et al., 2007; Hartikainen et al., 2009). However, much larger effects on VOC emissions appear to be caused by climatic factors such as temperature and drought.

One other potential role of biogenic VOCs in relation to ozone is their protective role as antioxidants within the leaf. At the leaf level, reaction of ozone with VOC emissions from plants may be sufficient to

reduce ozone uptake by stomata in some species, contributing to the protection of plants against ozone-induced damage (e.g. Fares et al., 2008). One of the postulated roles for isoprene production by plants, which is energetically expensive, is that it (and other reactive VOCs) may protect leaves from oxidative stress – not just from ozone exposure, but also from other abiotic and biotic causes (Kesselmeier and Staudt, 1999). However, direct reaction of VOCs with ozone within the leaf, in the apoplast, might not be sufficiently rapid to make a significant impact. The most likely reactant for ozone in the sub-stomatal cavity is ascorbate (Chameides, 1989; Conklin and Barth, 2004). Biogenic VOCs have much wider roles in ecosystem functioning (Laohawornkitkul et al., 2009), which may be affected by ozone. Recently Pinto et al. (2010) reviewed the effects of ozone on VOC emissions from plants and ecological interactions based on VOC signalling.

2.3 Ozone reduces CO₂ uptake by vegetation and damages leaf cells

Once inside the leaf, ozone can cause damage to the plant. Ozone interacts with the aqueous contents of the sub-stomatal pore and with adjoining cell membranes and walls, to form reactive oxygen species (ROS) such as hydrogen peroxide, superoxide, and hydroxyl radicals (reviewed in Fiscus et al., 2005). This induces a chain reaction whereby further ROS are formed within adjoining cells. Plants have a limited ability to detoxify ROS by “mopping up” or scavenging them via antioxidants such as ascorbic acid, flavonoids and phenolics or enzymes such as superoxide dismutase, catalase or peroxidases (Blokhina et al., 2003). Ozone tends to increase the total antioxidant capacity of plants (Gillespie et al., 2011). The ROS that remain unscavenged can affect physiological processes in the leaf such as stomatal conductance and photosynthesis. Ultimately, unscavenged ROS can cause a variety of leaf injury symptoms such as interveinal necrosis, and markings on the upper surfaces of leaves known as stipple, flecking, mottling, yellowing, bronzing, bleaching or tip-burn. Affected leaves commonly senesce, wither and fall off the plant early. These visible injuries are caused by free-radical induction of cell death and accelerated senescence and abscission, and evidence is growing that these processes are in part mediated by the plant hormones ethylene, jasmonic acid and salicylic acid (see Fiscus et al., 2005 and Kangasjärvi et al., 2005, for reviews).

A recent meta-analysis of ozone impacts on trees from temperate and boreal forests has shown that current ground-level ozone concentrations reduce the light-saturated rate of photosynthesis and stomatal conductance by 11 and 13% respectively compared to pre-industrial ozone concentrations (Wittig et al., 2007). In contrast to angiosperms (broadleaf trees), gymnosperms (including needle leaf trees as the largest group) were not significantly affected, which might be due to the generally lower stomatal conductance in gymnosperms compared to angiosperms. Younger trees (<4 years) were affected less than older trees. The decline in rubisco content and chlorophyll content may underlie significant reductions in photosynthetic capacity (Wittig et al., 2009). Decreased photosynthesis and ribulose-1,5-bisphosphate carboxylase/oxygenase (Rubisco) content are early symptoms of ozone exposure (Dizengremel, 2001; Long and Naidu 2002). These effects may be followed by accelerated senescence and decreased leaf area. Decreased C assimilation and altered C partitioning to stress-induced metabolic pathways may result in altered C allocation (Dizengremel, 2001). In a subsequent meta-analysis, Wittig et al. (2009) showed that current ambient ozone levels significantly reduce the total biomass of trees by 7% compared to trees exposed to pre-industrial ozone levels.

2.4 How ozone might ultimately reduce C sequestration

There are various potential routes by which ozone can ultimately result in a reduction in C sequestration in the plant and soil:

- Reduction in photosynthesis per unit leaf area (e.g. Morgan et al., 2003; Ainsworth et al., 2008; Wittig et al., 2007), either directly via effects on the photosynthetic machinery such as a reduction in Rubisco and chlorophyll content (Dizengremel, 2001; Fiscus et al., 2005; Wittig et al., 2009; Kobayakawa and Imai, 2011), and/or indirectly by closing the stomatal leaf pores (Torsehaugen et al., 1999; Evans et al., 2005; Overmyer et al., 2008; Wittig et al., 2009), resulting in a reduction in whole plant photosynthesis (see Section 2.3). On the other hand, mechanisms by which ozone might stimulate the opening of stomatal leaf pores transiently and hence affect whole plant photosynthesis and water balance have been discussed in more detail by Wilkinson and Davies (2010). Other studies have shown that ozone induces stomatal sluggishness, limiting CO₂ uptake and increasing water loss (Mills et al., 2009; Paoletti and Grulke, 2010).
- Reduction in whole plant photosynthesis due to ozone-induced damage of leaves, early senescence and abscission, and hence a reduction in total green leaf area and leaf area index (Morgan et al., 2003; Ainsworth et al., 2008; Wittig et al., 2009).
- As more C is required for secondary plant metabolism to detoxify ozone and/or repair ozone-induced cell damage (Betzelberger et al., 2010), less C will be available for plant growth (Dizengremel, 2001) and C allocation below-ground. In addition, the production of BVOCs is energetically expensive and might be stimulated by elevated ozone concentration, although contrasting results have been reported in the literature (see Section 2.2).
- Although increases in respiration per unit leaf area have been reported in response to high ozone exposure (e.g. Volin and Reich, 1996; Dizengremel, 2001), decreases (Wittig et al., 2009) have been reported too. However, in case of the latter, there seems to be a trend that the reduction in foliar respiration is less than the reduction in photosynthesis. Hence, relatively more of the C fixed at elevated ozone levels during daylight will be lost again to the atmosphere at night in response to ozone exposure.
- Another major effect of ozone on plants is to accelerate phenological development (e.g. Gelang et al., 2000). Maturity can be advanced by days or weeks, causing early flowering (Shi et al., 2009; Hayes et al., 2012), and early leaf senescence (Feng et al., 2011).
- A significant decline in the root to shoot ratio has been observed at elevated compared to either current ambient or pre-industrial ozone levels (Wittig et al., 2009). Hence, root biomass appears to be more sensitive to adverse effects of ozone than shoot biomass. In addition, increased ethylene formation, a common response to ozone pollution, tends to reduce root growth directly (Wilkinson and Davies, 2010). As a consequence, total C allocation to the soil is likely to be reduced.
- Open-air exposure to elevated ozone resulted in an increase in soil respiration (the sum of root and microbial respiration) due to a stimulation of fine root production in a beech and spruce forest (Nikolova et al., 2010). However, for spruce this was dependent on soil water availability as the ozone effect was observed in a wet year but not in a dry year. Microbial biomass and respiration were not significantly affected by elevated ozone in the aspen open-air exposure study (Larson et al., 2002). Mycorrhizal associations are generally enhanced by

elevated ozone (Gorissen et al., 1991; Scagel and Andersen, 1997; Matyssek et al., 2010). Elevated ozone can increase ectomycorrhizal colonization of host species, and alter the relative abundance of species within the ectomycorrhizal community despite having a generally negative impact on primary productivity (Andersen, 2003).

- Although open-air exposure to elevated ozone affected litter quality, this had little impact on microbial respiration. In contrast, litter quantity affected microbial respiration (Hillstrom et al., 2010). Litter from ponderosa pine trees with severe symptoms of ozone damage (chlorotic mottle) decomposed at the same rate as litter from adjacent trees with no visible chlorotic mottle (Fenn, 1991). Chapman et al. (2005) concluded that if changes in soil C cycling occur, they will most likely be brought about by changes in litter production. Holmes et al. (2006) showed that elevated ozone could reduce nitrogen availability in the soil via changes in litter production. As the effects of ozone on plant chemistry and ecological interactions are highly context- and species-specific, it difficult to identify general, global patterns.

2.5 Variation in sensitivity of species to ozone

Plant species vary in their sensitivity to ozone (e.g. Mills and Harmens et al., 2011), but the difference in sensitivity might be due primarily to an inherent difference in stomatal conductance (e.g. Wittig et al., 2007). On the other hand, species might inherently differ in their ozone detoxification potential (Frei et al., 2008, 2010a). There is even variation in ozone sensitivity between varieties or provenances of individual species (e.g. Mills and Harmens, 2011). Some of the variability in the susceptibility of wheat and rice varieties to ozone has been linked to the inherent rate of stomatal conductance in each. An ozone-sensitive biotype of white clover has also a higher stomatal conductance than the resistant biotype (Wilkinson et al., 2011). A role for ethylene in the genetic variability in ozone sensitivity in terms of visible injury has previously been demonstrated in *Arabidopsis* and poplar (Overmyer et al., 2003; see Wilkinson and Davies, 2009).

For species of (semi-)natural vegetation primarily occurring in grasslands and heathland, Hayes et al. (2007a) reviewed their ozone-sensitivity based on above-ground biomass production. Sensitive and insensitive species were identified as well as some species in which above-ground biomass was stimulated at higher ozone concentrations. Some relationships with plant physiological and ecological characteristics were identified. Based on the same database, Jones et al. (2007) developed a regression-based model for prediction above-ground biomass changes of individual species exposed to ozone based on their Ellenberg Indicator values for light and salinity. They showed that plant community sensitivity to ozone was primarily species driven. Mills et al. (2007b) developed the method further to predict ozone-sensitive plant communities. Dry grasslands, mesic grasslands, seasonally-wet grasslands and woodland fringes were identified as ozone-sensitive. Although alpine and subalpine grasslands and temperate shrub heathland were also identified as ozone-sensitive, these communities also contain a high proportion of species stimulated by ozone.

Various studies have shown that needle leaf trees are less sensitive to ozone than broad leaf trees (e.g. Wittig et al., 2009; Matyssek et al., 2010; Mills et al., 2011a). Matyssek et al. (2010) showed that enhanced ozone strongly reduces the C sink strength of adult beech as indicated by a 44% loss in stem productivity. In contrast, the sink strength of spruce was not affected and even displayed a slight increase in stem productivity (Pretzsch et al., 2010). This underlines once again the need to understand the species-specific nature of responses to ozone in order to assess impacts of ozone on C sequestration on forests. Rising ozone might not only decrease the productivity of forests, it might also give needle leaf trees a competitive advantage over broad leaved trees in mixed forests. Finally, adaptation of trees to oxidative stress such as ozone might occur in the natural environment (Paoletti, 2006). It should be noted that there is a lack of data on the impacts of ozone on tropical tree species.

2.6 Feedbacks: additional risks to vegetation

It can be concluded that ozone is likely to reduce overall C sequestration in terrestrial ecosystems. However, species-specific, ecosystem-specific and climate region-specific responses are to be expected. The negative impacts of ozone on plant functioning, growth and development is likely to result in a decline in ozone deposition to vegetation and hence removal of ozone from the atmosphere (Sitch et al., 2007). This will result in a positive feedback, enhancing radiative forcing due to both higher atmospheric ozone and CO₂ concentrations and therefore stimulating global warming. The contribution of ozone to global warming via its impacts on vegetation (i.e. a reduction in ozone deposition and C sequestration) might be as big as the direct impact of ozone as a greenhouse gas (Sitch et al., 2007).

3 Ozone effects in a changing climate

Harry Harmens, Sally Wilkinson, Gina Mills, Bill Davies

3.1 Introduction

For the next two decades, a warming of about 0.2°C per decade is projected for a range of emission scenarios. Depending on emission scenario, the predicted range of global warming by 2100 is on average ca. 2 – 4°C with warming predicted to be greater at higher northern latitudes (IPCC, 2007). Globally averaged mean water vapour and evaporation are projected to increase too. Increases in the amount of precipitation are very likely at high latitudes, while decreases are predicted in most subtropical land regions. A warmer future climate will also imply fewer frost days and increased summer dryness with greater risk of drought especially in the mid-continental areas. In terrestrial ecosystems there is evidence of earlier timing of spring events and poleward and upward shifts of plant and animal ranges, linked to global warming. Projected climatic changes will have an impact on the response of plants to ozone (Tausz et al., 2007).

Direct and indirect interactions between elevated ozone, elevated CO₂ and climate change will modify plant dynamics. In addition, ozone itself can modify the responses of plants to naturally occurring environmental stresses such as drought (e.g. Mills et al., 2009; Wilkinson and Davies, 2009, 2010). Ozone might alter the performance of herbivorous insect pests and plant pathogens, which themselves will be influenced by global warming. There is significant potential for the predicted changes in the climate to influence the response of vegetation to ozone through an effect on the rates of stomatal flux as the flux of ozone into the stomata is highly dependent on climatic conditions. Effects can be direct – e.g. temperature, CO₂ and humidity effects on stomatal conductance or indirect via an influence on soil water potential (SWP) and plant development (Harmens et al., 2007; Vandermeiren et al., 2009). In addition, climate change might affect the detoxification of ozone inside the leaves.

In this chapter we will review the interactions between ozone and other drivers of change in a future climate, i.e. elevated CO₂ concentrations, global warming, enhanced drought frequency and reduced nitrogen deposition.

3.2 Interactions between ozone and elevated CO₂: implications for C sequestration

It has previously been assumed and modelled that rising atmospheric CO₂ concentrations will be partially off-set by the effect of elevated CO₂ to increase the land C sink, through increased global plant biomass, which in turn removes more C from the atmosphere through increased photosynthetic C fixation (e.g. Jaggard et al. 2010, Bernacchi et al. 2006). Furthermore, because elevated CO₂ reduces the opening of stomatal pores (e.g. Wittig et al., 2007; Kim et al. 2010) and the leaf surface stomatal density (Lake and Woodward, 2008), elevated CO₂ can improve plant water use efficiency (e.g. Drake et al., 1997, Booker et al., 2004, but see Jaggard et al., 2010), and hence reduce plant susceptibility to stresses such as drought, high salinity and high vapour pressure deficit. This will tend to reduce plant injury and is likely to have a positive effect on the land C sink.

However, more recent field studies using Free Air CO₂ Enrichment (FACE) systems indicate that the positive effect of elevated CO₂ on C sequestration might be overestimated in previous studies using more controlled environments, smaller scale and/or shorter time periods (Long et al., 2005; Bernacchi et al., 2006). The actual increase in plant biomass, and its capacity to act as a C sink under elevated

CO₂ are smaller than predicted, partly due to accompanying ozone pollution, and partly due to accompanying increases in global temperature and drought frequency. Given that concentrations of CO₂ and ozone in our atmosphere are predicted to increase in parallel over the coming decades, through anthropogenic sources, it is implicit for understanding and predicting high CO₂ effects on the land C sink that we consider effects of both gases together (Sitch et al., 2007). There is an urgent need for more field-based, larger scale experiments where vegetation is exposed to multiple drivers of climate change for several years (at least one decade) to further test the above findings.

In Section 2.4 we discussed how ozone might affect C sequestration via different processes. Below we discuss in more detail how elevated CO₂ might affect the impact of ozone on the different processes (**Table 3.1**):

- It has often been reported in the literature that elevated CO₂ reduces stomatal conductance (Curtis and Wang, 1998; Drake et al., 1997; Morgan et al., 2003; Kim et al., 2010) and therefore the uptake of ozone and its damaging impacts on plants (Fiscus et al., 1997; McKee et al., 1997; Harmens et al., 2007). However, a recent stomatal flux modelling study based on a field experiment reported complex interactions between elevated CO₂ and ozone in northern hardwood forests: both gases stimulated leaf-level stomatal conductance whereas the combined gases did not affect leaf-level stomatal conductance in comparison with the control (Uddling et al., 2010).
- Compensatory interactions between ozone and CO₂ have also been demonstrated to occur directly at the level of the photosynthetic machinery (e.g. Kobayakawa and Imai, 2011), such that ozone and CO₂ effectively compensate for one another's effects on C fixation at the level of leaf physiology (e.g. Gray et al., 2010).
- Like ozone, elevated atmospheric CO₂ (Ludewig and Sonnewald, 2000) and increased C supply per se (Pourtau et al., 2004) can also accelerate leaf development and senescence. However, Gray et al. (2010) and Kontunen-Soppela et al. (2010a,b) amongst others, found that elevated CO₂ can delay senescence, and under some circumstances compensatory interactions between ozone and CO₂ can occur directly at the level of gene expression associated with senescence.
- With respect to BVOCs, much research has focussed on the emission of isoprene. In general, elevated CO₂ reduces isoprene emissions (Monson et al. 2007), thus opposing the large increase in isoprene emissions resulting from global warming-induced increases in net primary production. As described in Section 2.2, contrasting results have been reported regarding the impact of ozone on isoprene emissions.
- In contrast to ozone, elevated CO₂ tends to reduce total antioxidant capacity in soybean (Gillespie et al., 2011), which could result in a positive effect on the land C sink via reduced C allocation into antioxidant production. However, experimental data are inconclusive with respect to a general ameliorating effect of CO₂-enrichment on ozone-induced oxidative stress via changes in the antioxidant status of leaves (Vandermeiren et al., 2009).
- Contrasting responses have been reported for the impact of elevated CO₂ on C allocation to roots, as increases, decreases and no changes in root-shoot ratio have been found (e.g. Rogers et al., 1996; Maroco et al., 2002; Kimball et al., 2007). Competition between species is likely to affect the response of individual species (Kozovits et al., 2005). Although high CO₂ alleviates the effect of ozone in reducing below ground root biomass in woody species, this seems not to be the case in herbaceous species (Wang and Taub, 2010). Wang and

Taub (2010) concluded that elevated CO₂ has less pronounced effects on the root mass fraction than other environmental factors.

- Previously it has been suggested that elevated CO₂-induced increases in microbial biomass will lead to greater soil C storage (e.g. Van Ginkel et al., 1997). However, more recently it has been demonstrated that elevated CO₂-induced increases in the amount of soil organic matter in less stable pools such as microbial biomass, reducing soil capacity as a sink for C by making less C available to the more stable pools (Hofmockel et al., 2011; Zak et al., 1993; Carney et al., 2007; Cheng et al., 2011). Ozone increased the amount of C entering more stable pools in the study by Hofmockel et al (2011) in a northern European hardwood forest, but not in other studies (Esperschütz et al., 2009; Cheng et al., 2011). In a recent meta-analysis, Dieleman et al. (2010) showed that elevated CO₂ induces a C allocation shift towards below-ground biomass compartments. However, the increased soil C inputs were offset by increased heterotrophic respiration, such that soil C content was not affected by elevated CO₂. Soil nitrogen concentration strongly interacted with CO₂ fumigation: the effect of elevated CO₂ on fine root biomass and -production and on microbial activity increased with increasing soil nitrogen concentration, while the effect on soil C content decreased with increasing soil nitrogen concentration. These results suggest that both plant growth and microbial activity responses to elevated CO₂ are modulated by nitrogen availability, and that it is essential to account for soil nitrogen concentration in C cycling analyses.

Table 3.1 Ozone and CO₂ impacts on plant physiology and other processes are often in opposite directions.

Parameter	Ozone	CO ₂
Photosynthesis	-	+
Stomatal conductance*	-/+	-
Leaf area index	-	+
Ratio photosynthesis:respiration	-	+
Phenology	+	-/+
Ratio root:shoot biomass	-	-/+
Isoprene emissions**	-/+	-
Soil respiration	-/+	+

* '-' reduces, '+' enhances susceptibility to drought stress

** warming enhances isoprene emissions

In summary, elevated ozone and CO₂ often affect plant physiology and soil processes in opposite directions. Hence, the direction of response is finely balanced dependent on the relative concentrations of both gases, and the possibility for greater than additive effects under field conditions.

3.3 Ozone impacts on C sequestration in a warmer climate

The complexity of the interactions between the factors involved in climate change is well illustrated by consideration of the impacts of global warming on the canopy uptake of ozone. When considered as a single factor, increased temperature is likely to increase stomatal uptake of ozone providing the optimum temperature for stomatal conductance has not been reached (Emberson et al., 2000a), e.g. in temperate climates. However, the response to warming will also be affected by the following indirect effects of increased warming: added stimulation of tropospheric ozone formation, an increase

in vapour pressure deficit, a decrease in soil water potential (soils will dry out faster due to enhanced soil evaporation and enhanced canopy transpiration), and earlier and enhanced plant development, resulting in a forward shift of the period within the year when plants are absorbing ozone. Thus, the overall impact of warming on the canopy flux of ozone is difficult to predict and will depend on the severity and timing (e.g. summer or winter) of warming and changes in precipitation together with any changes in seasonal patterns in the occurrence of peak episodes of ozone. Little is known about the impacts of a few degrees rise in temperature on the antioxidant status of leaves and thus on ozone detoxification.

At the same time global warming will also affect photosynthesis and plant and soil respiration. The impact of warming on C sequestration will depend on the change in balance between plant photosynthesis and ecosystem respiration. Few studies have been conducted on vegetation responses to the combined impacts of ozone and warming. Kasurinen et al. (2012) showed that elevated temperature increased above- and below-ground growth and soil respiration rates in silver birch. However, for some of these variables the temperature effect was modified by tree genotype and prevailing atmospheric ozone concentration. Although warming has the potential to increase silver birch growth and hence C accumulation in tree biomass, the final magnitude of this C sink strength is partly counteracted by temperature-induced increase in soil respiration rates and simultaneous ozone stress. Silver birch populations' response to climate change will also largely depend on their genotype composition (Kasurinen et al., 2012).

3.4 Interactions between ozone and drought: implications for C sequestration

Since ozone episodes frequently co-occur with climatic conditions associated with drought and an increased frequency of drought is predicted for the coming decades (IPCC, 2007), it is important to understand how vegetation will respond to the combined stresses of ozone and drought in order to predict future impacts on C sequestration. It has been widely reported that drought-induced stomatal closure will limit ozone uptake and damage (e.g. Fuhrer, 2009, Fagnano et al., 2009). However, recent studies have shown that drought does not always reduce ozone-induced damage to plants in sensitive species (Mills et al., 2009; Wilkinson and Davies 2009, 2010; Wagg et al. 2012), and that the genetic variability in ozone sensitivity may be related to the extent to which ozone reduces the sensitivity of stomatal closure to soil drying (see below). Other studies have also shown that the expected protective effect of drought on deleterious plant responses to ozone did not occur (e.g. Heggestadt et al., 1985; Robinson et al., 1998; McLaughlin et al., 2007).

In drying soil, stomata of some species close much less sensitively in ozone-polluted air, and ozone can even open stomata under well-watered conditions in some cases (Mills et al., 2009, Wilkinson and Davies 2009, 2010). The extent of this effect is maybe genetically determined (Wilkinson et al., 2011). Stomata in sensitive genotypes close less sensitively in response to drought signals such as abscisic acid accumulation generated by the plants, because of the concomitant effect of ozone to increase stress ethylene emission, which interferes with the stomatal closure response to abscisic acid. This reduced stomatal closing response to ozone will also directly increase plant water loss, and therefore increasing vulnerability to the drought episode (particularly when combined with a reduced root biomass – Grantz et al., 2006). This might eventually cause secondary reductions in C sequestration (Wilkinson and Davies 2009, 2010), particularly if the vulnerable plants begin to experience additional/subsequent stresses such as wind, biotic attack, high light/VPD or flood/storm conditions (Wilkinson and Davies, 2010). A growing number of species exhibit ozone-induced stomatal opening either in the presence or absence of soil drying, that is genotype-dependent (Wilkinson et al., 2011). However, the ozone-induced initial increases in both stomatal aperture and

leaf surface area growth are thought to be temporary. They are not sustainable as leaf tissue will eventually become water-stressed, eventually resulting in a reduction in C sequestration.

In the dry year of 2003, prolonged water shortage rather than ozone stress limited both radial and whole-stem volume increment of beech trees (Matyssek et al., 2010). Drought can also override the stimulating ozone effects on fine-root dynamics and soil respiration in mature beech and spruce forests (Nikolova et al., 2010). At the leaf level, the impact of ozone was reduced because from early summer drought-driven stomatal closure pre-empted ozone-driven effects (L  w et al., 2006).

3.5 Interactions between ozone and nitrogen deposition: implications for C sequestration

As for ozone, nitrogen (N) deposition to vegetation has increased since the industrial revolution. However, the implementation of air pollution abatement strategies in Europe in recent decades have resulted in a steady decline in N emissions since 1990 (Sutton et al., 2011; EMEP, 2011) and they are predicted to decline further in the future. However, global emissions of ammonia have risen again since 2000 and are predicted to rise further by 2050 (Dentener et al., 2005). De Vries and Posch (2011) showed in a modelling study that N deposition was the main driver for increased forest growth and C sequestration in temperate forests in the past, but predicted that climate change will be the main driver in the future. The results of the various studies are generally in agreement and show that aboveground accumulation of C in forests is within the range 15 - 40 kg C kg⁻¹ N (De Vries and Posch, 2011, and references therein). However, the stimulating effect of climate change in the future will be limited by the future reduction of N deposition and the availability of other nutrients for forest growth. In addition, past and future impacts of changes in ground-level ozone concentrations were not taken into account in the modelling exercise. On the other hand, Nadelhoffer et al. (1999) concluded that N deposition makes only a minor contribution to C sequestration in temperate forests.

Relatively few studies have investigated the impacts of both ozone and N on vegetation. Evidence suggests that ozone and N can have both synergistic and antagonistic effects on species and ecosystem processes, and that they may interact in unpredictable ways to affect plant communities (Harmens et al., 2006). N alleviated the negative effect of ozone on root starch concentrations in *Picea abies*, but otherwise no significant interactive impacts were observed on growth parameters (Thomas et al., 2005). Handley and Grulke (2008) showed that increasing N ameliorated the negative impact of ozone on black oak. Reports on the combined impacts of ozone and N on non-tree species suggest that effects are complex and dependent on several factors, not least inter-specific differences in life-strategy and functional type. Whitfield et al. (1998) found that *Plantago major* plants grown in controlled conditions were more sensitive to ozone at low N. Others reported no significant interactions or antagonistic effects (e.g. Cardoso-Vilhena and Barnes, 2001). In addition, interactive impacts can vary for above- and belowground plant parts (Jones et al., 2010). Phenotypic plasticity in growth rate and leaf area (Bassin et al., 2007) might influence the response to both ozone and N pollution, however, the responses are likely to be affected by other environmental factors such as temperature and availability of other nutrients (Bassin et al., 2009). Furthermore, the ozone sensitivity of plant species does not seem to be affected by their Ellenberg nutrient score (Jones et al., 2007).

3.6 Conclusions

The impacts of ozone on C sequestration in terrestrial ecosystems in Europe in the future will depend on the interaction and magnitude of other environmental and climate changes such as rising temperature, increased drought frequency, enhanced atmospheric CO₂ concentration and reduced N deposition. Ecosystems are inherently complex, and for any one parameter of functioning such as forest growth, there are multitudes of driving factors (Lindner et al., 2010). So far, the majority of

studies on the impacts of drivers of change have been relatively short term, small scale and in more or less controlled climate conditions. Large scale and long-term free air exposure studies have been limited so far due to the high cost of such studies. Moreover, relatively few studies have investigated the interactive impacts of two or more drivers of change. The outcome of such studies often indicate complex interactions between drivers of change and non-linearity in responses. On average, both positive and negative global change impacts on the biosphere might be dampened more than previously assumed (Long et al., 2005; Leuzinger, 2011). There is an urgent need for more field-based, larger scale experiments where vegetation is exposed to multiple drivers of climate change for several years (at least one decade) to further investigate the overall impact of a combination of drivers of change on terrestrial ecosystems.

Modelling studies to predict future impacts of change should also be based on a multifactorial approach, including changes in ground-level ozone concentrations. So far, the impacts of ozone on vegetation and feedbacks to the climate have hardly been considered in global climate modelling (but see Sitch et al., 2007; Collins et al., 2010; Huntingford et al., 2011). Predictions for the future are also complicated by unknown changes in land use and management which also have a major impact on the future C sequestration potential of terrestrial ecosystems. Furthermore, the predicted increase in extreme weather (IPCC, 2007) makes it even harder to predict future impacts. For example, the Europe-wide heat and drought seen in 2003 was shown to have caused a 30% reduction in primary productivity, resulting in a net C release that equated to four years' worth of sequestered C, inducing a 0.5 Pg C yr^{-1} net output of C dioxide (Ciais et al., 2005).

4 Impacts of current and future ozone concentrations on carbon uptake and storage in trees across Europe: application of DO₃SE

Patrick Büker, Lisa Emberson, Richard Falk, Alan Briolat, Steve Cinderby, Howard Cambridge, Harry Harmens, Gina Mills, David Norris, David Simpson

4.1 Introduction

Within the terrestrial biosphere, forest ecosystems have the greatest C sink capacity of any vegetation type (Janssens et al., 2003; Luyssaert et al., 2010) and indeed hold the largest amount of biomass C, totalling 50% of all terrestrial C (Körner et al., 2005). Many experimental studies have shown that current baseline levels of tropospheric ozone induce biomass reductions in trees (Wittig et al., 2009 and references therein). This has major repercussions for the global C cycle and climate change policy as the terrestrial biosphere removes approximately a third of all present day anthropogenic CO₂ emissions (Felzer et al., 2004; Canadell et al., 2007; Royal Society, 2008). Europe (including eastern European countries such as the Russian Federation) alone holds 25% of the world's total forests, totalling 1.02 billion hectares of forested area (Forest Europe, 2011). European forests are predicted to currently sequester 0.11 Pg C yr⁻¹, which is approximately 10% of the European emissions (De Vries & Posch, 2011). For the EU-27 countries, Norway and Switzerland, this translates to a total C biomass stock of 10445 Mt C, a figure which includes both above-ground and below-ground (roots) biomass (Forest Europe, 2011). The largest stocks exist in northern and central European countries (Forest Europe, 2011), meaning that temperate and boreal forests of these regions comprise the most important C sink in the EU-27 (Wittig et al., 2009; see also Chapter 6). The locations most at risk from biomass C reductions are regions with both high effective ozone 'uptake' by trees and high biomass C stocks.

Table 4.1 Summary of the C cycle in forests in the EU-25 (Luyssaert et al., 2010). Fluxes are in Tg C yr⁻¹ over a forest area of 1.46×10^6 km². Heterotrophic respiration (decomposition) was estimated as the residual term to make the balance close, the observed heterotrophic respiration is 600 ± 45 Tg C yr⁻¹. The biomass + soil C sink was estimated to be almost 40% (40 Tg C yr⁻¹) due to land use change. For comparison, fossil fuel burning emits 1060 ± 100 Tg C yr⁻¹.

Component flux	Influx	Efflux	Source/sink
Net primary productivity (NPP)	756 ± 98		
Wood harvest		92 ± 16	
Decomposition on site		533 ± 105	
Wood products decomposition		87 ± 16	
Fires		7 ± 2	
River C flux		15 ± 6	
Biomass sink*			80 ± 15
Soil C sink			29 ± 18

* Including wood-based products sink of 5 ± 3 Tg C yr⁻¹.

Inventory-based assessments suggest that 29(±15)% of the net biome productivity (NBP) is sequestered in the forest soil, but large uncertainty remains concerning the drivers and future of the soil organic C (Luyssaert et al., 2010). The remaining 71(±15)% of the NBP is realized as woody biomass. In the EU-25, the relatively large forest NBP is thought to be the result of a sustained

difference between net primary productivity (NPP), which increased during the past decades, and C losses primarily by harvest and heterotrophic respiration, which increased less over the same period. A summary of the C fluxes in European forests (EU25) based on eddy covariance measurement is provided in **Table 4.1** (Luyssaert et al., 2010). During the past 50 years, NPP of forests in the EU-25 increased by a factor of 1.7 and the biomass stock per area of forest increased in parallel by a factor 1.8 (Ciais et al., 2008). Moreover, forest C stocks increased everywhere in Europe linearly with NPP. Future NPP is expected to further increase above current NPP levels owing to predicted increases in temperature (Meehl et al., 2007) and CO₂ concentration (Norby et al., 2005; Meehl et al., 2007). Expected increase in NPP could, however, be offset by progressive limitations of nutrients (Dentener et al., 2006; Gill et al., 2006; Luo, 2007), water stress related to the predicted decreased growing season precipitation (Meehl et al., 2007), climate change-induced changes in species composition (Cramer et al., 2001; Parmesan and Yohe, 2003; Jump et al., 2006), increasing ozone concentrations (Sitch et al., 2007), increasing frequency of insect outbreaks (Percy et al., 2002; Logan et al., 2003), increased frequency and intensity in forest fires owing to changes in species composition and climate (Westerling et al., 2006; Bond-Lamberty et al., 2007) or increasing storm damage because of increased storm intensity in the temperate zone (Meehl et al., 2007) in combination with the increased cultivation of tree species outside their natural range (Schelhaas et al., 2003). Recently the role of climate variability and extreme weather conditions such as the 2003 heat wave has been identified as a factor that may strongly alter responses of productivity to mean climate change (Ciais et al., 2005; Reichstein et al., 2007). Hence, it remains uncertain whether NPP will increase or decrease in the next decades.

The aims of the modelling case study described in this chapter were to:

- Estimate the ozone induced reduction of C storage in living biomass of forests under present (year 2000) and future (2040) climate/meteorology and emission conditions.
- To assess the influence of climate and species-specific flux and flux-response parameterisations on ozone-induced reduction of C storage in living biomass estimates for present meteorological conditions (year 2000).
- To assess the influence of soil water stress on ozone-induced reduction of C storage in living biomass by activating and deactivating the DO₃SE soil water module for the year 2000.
- To compare the spatial distribution of AOT40 and POD_Y estimates of ozone-induced reduction of C storage in living biomass of forests.

4.2 Methodology

Many process-based, C flux models that simulate the impact of a dynamic, multi-factor environment on plant growth use the AOT40 metric as ozone input data (e.g. Felzer et al., 2004; Ren et al., 2007). Recently, Sitch et al. (2007) used the biologically more relevant stomatal ozone flux metric as a measure of ozone dose (see Chapter 5). However, Sitch et al. (2007) used previously developed stomatal flux-response relationships (Karlsson et al., 2004; Pleijel et al., 2004), whereas in the current study we applied recently revised flux-response relationships (Mills et al., 2011a) and for a broader range of species when using climate-specific parameterisations. Biomass reduction was related to hourly estimates of stomatal ozone flux, accumulated over a species growing season (POD_Y (Phytotoxic Ozone Dose above a threshold flux of Y nmol m⁻² s⁻¹; LRTAP Convention, 2010; Mills et al., 2011a).

4.2.1 Input data

For this modelling case study the following input data were used:

- Atmospheric ozone concentration and associated meteorological data: i) ozone and meteorological data provided by EMEP for the year 2000 (Simpson, pers. comm.) and ii) ozone and climate data provided by the Rossby Centre regional Atmospheric climate model (RCA3, referred to as RCA from now on; Kjellström et al., 2005) for current (2000 to 2009) and future (2040 to 2049) climate years.
- Land cover data to identify the distribution of forests tree species: i) for EMEP data the species-specific JRC land cover data (<http://forest.jrc.ec.europa.eu/distribution>) and for ii) RCA data the LRTAP Convention harmonised land cover data (Cinderby et al., 2007) were used.
- Forest C stock data were derived from the European forests inventory dataset (Forest Europe, 2011), an inventory conducted as part of Europe's Kyoto commitment providing country based estimates of C stock.

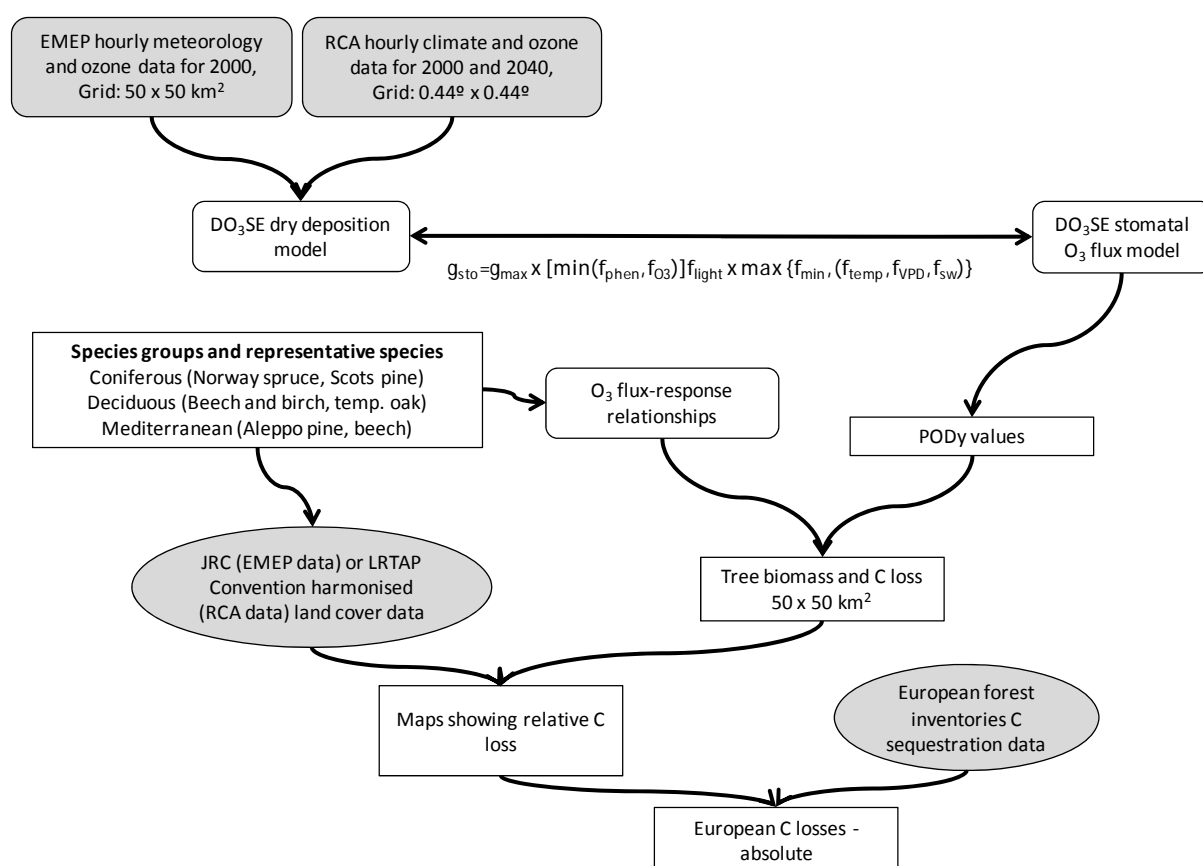


Figure 4.1 The combination of different datasets used within this case modelling study for forests.

The data were aggregated to the EMEP grid (50 km x 50 km) and RCA grid (0.44° x 0.44°). **Figure 4.1** shows how the different input datasets (indicated by the grey shaded boxes) were combined, using the EMEP year 2000 ozone and meteorology and RCA 2000 and 2040 ozone and climate data as an example. The meteorological data were used both to generate the ozone concentration fields (within the EMEP photochemical model) and also to drive the DO₃SE modelled ozone dry deposition and stomatal flux (F_{st}) estimates (Emberson et al., 2000a,b, 2001, 2007; Büker et al., 2011).

Application of the DO₃SE dry deposition model transformed EMEP ozone concentrations provided at an approximate height of 50m above the surface to vegetation canopy height, assumed to be 20m for forests. The DO₃SE model F_{st} values were then used to estimate POD_Y values for a number of different tree species in various climatic regions in Europe (LRTAP Convention, 2010). The POD_Y values were used to estimate relative biomass and subsequent steady state C reductions in the living biomass of forests by using appropriate flux-response relationships (LRTAP Convention, 2010). These relative values were then used to estimate actual or absolute European C stock reductions in the living biomass of forests due to ozone based on the location of each receptor, defined according to either the JRC land cover map (EMEP input meteorological and ozone data) or the LRTAP Convention harmonised land cover map (RCA input climate and ozone data), and data describing existing absolute C stocks.

Given that ozone stomatal flux parameterisations and flux-response relationships only exist for a rather limited number of European forest species, the ultimate outcome of this modelling and mapping exercise was to estimate a 'ballpark' figure for C reduction in the living biomass from forests due to ozone. However, the availability of species- and climate-specific flux as well as flux-response parameterisations and new methods to estimate the effect of soil moisture on F_{st} of forest trees allows the influence of these factors to be assessed in relation to C reduction estimates. This enables an assessment of the variability and uncertainty in the broader scale modelling approach. To this end, a two-fold modelling approach has been used: Firstly, to investigate ozone impacts in a future climate, DO₃SE model runs were performed using modelled RCA time series of **climate** data representing present (2000) and future (2040) climate conditions. Secondly, to test a) the sensitivity of the effect of different parameterisations and flux-response relationships and b) the influence of soil water on the final C sequestration value, additional DO₃SE model runs were performed using EMEP **meteorology** data for the year 2000 only.

RCA model: ozone and climate data

Climate data were generated using the RCA model (RCA3; Kjellström et al., 2005), which is a regional climate model that receives relevant climate input parameters from a Global Circulation Model (GCM). RCA includes a description of the atmosphere and its interaction with the land surface and generates climate data for a 0.44 degree grid (i.e. cell size: 0.44° x 0.44°) that covers the whole of Europe. For this study, 10-year hourly time-series data for 2000 to 2009 and 2040 to 2049 were generated, using GCM simulations based on concentrations of atmospheric CO₂ and other climate-relevant gases for the year 2000 and projected for the year 2040. These 10-year time-series were required because only a succession of 10 years of data was believed to be sufficiently statistically robust to represent the current (i.e. year 2000) and future (i.e. year 2040) climate.

The 2000 scenario runs used emissions from the year 2000, together with RCA meteorology for the years 2000-2009. Model runs were performed for each year and resulting AOT40 and POD_Y values were then averaged to get representative hourly AOT40 and POD_Y values for the year 2000. It should be noted that this is climate model meteorology, not real meteorology, and should be regarded as a statistical collection meant to represent near-current conditions. The 2040 scenario runs used the GEA-LOW-CLE emissions generated by IIASA for the year 2050 (<http://cityzen-project.eu>), together with RCA meteorology for 2040-2049. Thus, both emissions and meteorology were changed. The GEA-LOW-CLE emission scenario is based on the illustrative scenario of the GEA Efficiency pathway group in terms of energy demand and use, and the implementation of a stringent climate policy corresponding to a maximum of 2 °C rise in global temperature target. In addition, this scenario assumes global implementation of extremely stringent pollution policies (SLE) until 2030. These stringent air quality control strategies are much more aggressive than the currently planned legislations, but are still lower than the so called Maximum Feasible Reduction (MFR) which describes the technological frontier in terms of possible air quality control strategies by 2030.

EMEP model: ozone and meteorological data

Atmospheric ozone concentrations for the year 2000 were modelled on an hourly basis by the EMEP (European Monitoring and Evaluation Programme) photo-oxidant model (Simpson et al., 2003). The model uses both 3-hourly resolution meteorological data from the High Resolution Limited Area Model (HIRLAM) and anthropogenic emissions data that are predominantly derived from official national estimates (Simpson et al., 2007). The EMEP model simulates atmospheric events such as the emission of primary air pollutants (i.e. ozone and ozone precursors), transformative processes (both chemical and physical) and loss processes to land surfaces, such as vegetation (Tuovinen et al., 2007). The EMEP model comprises 20 vertical layers and provides output at a horizontal resolution of 50 x 50 km², at a reference height of 50m (Simpson et al., 2003, 2007).

DO₃SE model input data

Ozone and meteorological data from both the RCA and EMEP models were supplied for a region of Europe that included the EU-27, Norway and Switzerland, according to the respective grids of the modelling domains. The ozone data supplied at the 50m reference height with the off-line DO₃SE model (Emberson et al., 2000a,b, 2001, 2007; B  ker et al., 2011) were transformed to ozone concentrations to the canopy height (20m). In order to achieve this, the DO₃SE model requires certain meteorological data, which are described in **Table 4.2**.

Land cover data

For EMEP climate and ozone data, the species-specific JRC land cover data (<http://forest.jrc.ec.europa.eu/distribution>) had to be used to enable the application of climate region-specific parameterisations, whereas for RCA climate and ozone data the UNECE LRTAP Convention harmonised land cover data (Cinderby et al., 2007) was used, as only information on the distribution of deciduous, coniferous and mixed forests was required as described in the harmonised land cover data. As such, the JRC land cover data were overlain with the EMEP 50 x 50 km² resolved grids and the LRTAP Convention harmonised land cover data were overlain with the RCA 0.44  x 0.44   resolved grids to produce species specific and generic land cover data aggregated to each of the model domains grid squares. It should be noted that the JRC land cover data does not provide data for Portugal, Cyprus and part of Italy (e.g. Sardinia).

Table 4.2 Meteorological and ozone data required to run the DO₃SE model

Input data	Height (m)	Units
Ozone concentration (O ₃)	Reference height and surface canopy height	ppb
Horizontal wind speed (u)	Reference height and surface canopy height	m s ⁻¹
Air pressure (p)	surface	N m ⁻²
Global radiation OR Photon Photosynthetic Flux Density (PPFD)	Top of canopy Top of canopy	W m ⁻² ��mol m ⁻² s ⁻¹
Air temperature (T _{air}) / Leaf temperature (T _{leaf})	surface	K
Vapour Pressure Deficit (VPD)	Surface	kPa
Precipitation (Pr)	Ground	mm
Site/Grid specific variables	Character	Units
Latitude and Longitude (lat & long)	-	�, �
Elevation (e)	-	m a.s.l.
Target canopy height (tgt)	-	m
Soil texture	coarse / medium / fine	-

Forest inventory C stock data

Biomass C totals per country were used for the year 2000 as determined by the 'State of Europe's Forests 2011' report (Forest Europe, 2011). The data was sourced from national reports on quantitative indicators that were directly provided by countries through the national enquiry (Forest Europe, 2011) through Kyoto Protocol biomass C measuring procedures (IPCC, 2006) and represents the whole tree C stock (above and below ground). **Table 4.3** shows the forest C stock for the different climate regions (LRTAP Convention, 2010) in Europe.

The C stock held within each EMEP and RCA grid was estimated assuming an even distribution of C stock across the forested area of the country. The C stock held within each species and land cover class within each grid was similarly divided assuming an area weighted distribution (**Figure 4.2**).

Table 4.3 Average C stock in forest biomass per EMEP and RCA grid (Mt C) and total C stock in forest biomass (Mt C) for European climate regions, based on country resolved forest inventory data for the year 2000 (Forest Europe, 2011). NE = Northern Europe, ACE = Atlantic Central Europe, CCE = Continental Central Europe, ME = Mediterranean Europe (LRTAP Convention, 2010).

	Europe	NE	ACE	CCE	ME
Forest C stock (Mt C), per EMEP grid	3.86	4.43	1.00	5.49	2.10
Forest C stock (Mt C), per RCA grid	3.81	3.98	1.03	5.94	2.02
Forest C stock (Mt C), total	9233.6	2881.7	231.4	4778.4	1342.1

4.2.2 The DO₃SE ozone dry deposition model

DO₃SE is a soil-vegetation-atmosphere-transport model that has been specifically designed to estimate the total and stomatal deposition of ozone to European vegetation (Emberson et al., 2001). Ozone concentrations at canopy height (20 m for forests) were used to calculate the stomatal ozone flux (F_{st}). The F_{st} calculation assumes that the ozone concentrations at the top of the canopy provide a reasonable estimate of the concentration at the upper surface of the laminar layer near the sunlit upper canopy leaves (LRTAP Convention, 2010). The atmospheric resistance to ozone transfer is estimated assuming a stable atmosphere. Further details of the methods used to estimate the ozone transformations are provided in Emberson et al. (2000a,b; 2001; 2007) and Simpson et al. (2003). To estimate leaf/needle stomatal conductance (g_{sto}), a key component of the F_{st} calculation, the DO₃SE model currently employs a multiplicative algorithm, based on that first developed by Jarvis (1976), modified for ozone flux estimates (Emberson et al., 2000a,b; 2001; 2007):

$$g_{sto} = g_{max} f_{phen} f_{light} \max\{f_{min}, f_T f_{VPD} f_{SW}\} \quad [1]$$

where the species-specific maximum g_{sto} (g_{max}) is modified by functions (scaled from 0 to 1) to account for g_{sto} variation with leaf/needle age over the course of the growing season (f_{phen}) and the functions f_{light}, f_T, f_{VPD} and f_{SW} relating g_{sto} to irradiance, temperature, vapour pressure deficit and soil water, respectively; f_{SW} can either be related to soil water potentials (f_{SWP}) or plant available soil water expressed in volumetric terms (f_{PAW}); f_{min} is the minimum daylight g_{sto} under field conditions, expressed as a fraction of g_{max}.

This stomatal component of the DO₃SE model is the primary determinant of F_{st}; the plants internal ozone detoxification capacity determines the fraction of this F_{st} that is effective in causing plant

damage. The DO₃SE F_{st} model has been used extensively to analyse experimental data describing ozone fumigation and plant response (e.g. Pleijel et al. 2007; Karlsson et al. 2007). This analysis has allowed the establishment of a number of flux-response relationships for a variety of different species (LRTAP Convention, 2010; Mills et al., 2011a). Since the methods used to derive the flux-response relationships are consistent with the modelling of dry deposition of ozone, it is to be expected that the DO₃SE model can be used with some degree of accuracy to estimate damage due to elevated ozone exposures as is done in this study. DO₃SE was used here to provide estimates of ozone flux for key tree species for climate model runs and 2000 meteorology model runs.

The DO₃SE model has recently been updated by including a method to estimate soil moisture status and its influence on g_{sto} for a variety of forest tree species (Büker et al., 2011). This soil moisture module uses the Penman-Monteith energy balance method (Monteith, 1965) to drive water cycling through the soil-plant-atmosphere system and empirical data describing g_{sto} relationships with pre-dawn leaf water status to estimate the biological control of transpiration. The impact of soil moisture on flux within this project was tested by activating and deactivating this soil moisture module.

Stomatal ozone flux modelling

F_{st} (nmol O₃ m⁻² PLA s⁻¹) is calculated according to eq. (2) which accounts for deposition to the cuticle through incorporation of the leaf surface resistance (r_c) and boundary layer resistance (r_b) terms:

$$F_{st} = c(z_1) * g_{sto} * \frac{r_c}{r_b + r_c} \quad [2]$$

where c(z₁) is the concentration of ozone at the top of the canopy (nmol m⁻³) at height z₁ (m), g_{sto} is the stomatal conductance in m s⁻¹, r_b is the leaf quasi-laminar resistance and r_c the leaf surface resistance (s m⁻¹). For further details on the resistance scheme, the interested reader is referred to Emberson et al. (2000, a,b; 2007) and LRTAP Convention (2010).

The accumulated F_{st} above an ozone stomatal flux threshold of Y nmol m⁻² s⁻¹ (POD_Y) is calculated as described in eq. (3) with the accumulation estimated using hourly F_{st} values over the entire growing season:

$$POD_Y = \sum_{i=1}^n [F_{sti} - Y] \text{ for } F_{sti} \geq Y \text{ nmol m}^{-2} \text{ PLA s}^{-1} \quad [3]$$

where F_{sti} is the hourly mean ozone flux in nmol O₃ m⁻² PLA s⁻¹, and n is the number of hours within the accumulation period. The threshold Y is here taken to equal 1 or 1.6 nmol O₃ m⁻² PLA s⁻¹ for consistency with the LRTAP Convention Modelling and Mapping Manual recommendations (LRTAP Convention, 2010; Mills et al., 2011a) and Karlsson et al. (2007), respectively.

Parameterisation of the DO₃SE model

Table 4.4 lists the species- and climate-specific parameterisations that were applied for the climate and meteorology model runs; those parameterisations that were used both for climate AND meteorology model runs are shaded in grey. Model runs using the DO₃SE model were only carried out for those tree species for which dose-response relationships were available (Karlsson et al., 2007; LRTAP Convention, 2010; Mills et al., 2011a). If not mentioned otherwise, model runs were performed taking into account the effect of soil moisture on g_{sto} (i.e. with activated soil moisture deficit module of DO₃SE model; Büker et al., 2011).

For the **climate** data model runs using the RCA data representing the years 2000 and 2040, the following simplified parameterisations were employed (Table 4.4):

- European deciduous trees were represented by the combined parameterisation for beech (*Fagus sylvatica*) and birch (*Betula pendula*) grown in Continental Central Europe (LRTAP Convention, 2010);
- European coniferous trees were represented by the parameterisation for Norway spruce (*Picea abies*) grown in Continental Central Europe (LRTAP Convention, 2010).

For the standard meteorology model runs using EMEP data for the year 2000, a similar parameterisation as that one described above for RCA data was used to enable a comparison between these two modelling approaches:

- European deciduous trees were represented by the 'generic deciduous' parameterisation as suggested by the LRTAP Convention (2010);
- In the absence of a 'generic coniferous' parameterisation, European coniferous trees were represented by the parameterisation for Norway spruce (*Picea abies*) grown in Continental Central Europe (LRTAP Convention, 2010; **Table 4.4**).

To test the sensitivity of the effect of climate region specific (CRS) parameterisations as compared to the above outlined standard parameterisation, **meteorology** model runs using EMEP data for the year 2000 were carried out by applying all available forest tree parameterisations as described in LRTAP Convention (2010) and Karlsson et al. (2007; **Table 4.5**):

- European deciduous trees were represented by parameterisation for i) birch (*Betula pendula*) grown in Northern Europe, ii) beech (*Fagus sylvatica*) and temperate oak (*Quercus petraea* and *Q. robur*) grown in Atlantic Central Europe, iii) beech and birch grown in Continental Central Europe and iv) beech grown in Mediterranean Europe (Karlsson et al., 2007; LRTAP Convention, 2010);
- European coniferous trees were represented by parameterisation for i) Norway spruce (*Picea abies*) grown in Northern Europe, ii) Scots pine (*Pinus sylvestris*) grown in Atlantic Central Europe, iii) Norway spruce grown in Continental Central Europe and iv) Aleppo pine (*Pinus halepensis*) grown in Mediterranean Europe (Karlsson et al., 2007; LRTAP Convention, 2010);

To test the influence of soil water on the final C sequestration value, additional **meteorology** model runs using EMEP data for the year 2000 were performed. The parameterisation was the same as for the standard model runs (see above), with the difference that the soil moisture deficit (SMD) module (Büker et al., 2011) of the DO₃SE model was turned off, assuming no drought effects on stomatal conductance and hence flux. This scenario is denoted SMD_{off} in the following.

4.2.3 Stomatal ozone-flux response relationships

Estimates of ozone induced relative biomass reduction (RBL) for forest trees were made by relating POD_Y values, estimated across Europe for different forest species and climate-specific parameterisations, to the appropriate dose-response relationships (Karlsson et al., 2007; LRTAP, 2010; Mills et al., 2011a) as listed in **Table 4.5**. To allow comparisons between the POD_Y and AOT40 approaches, ozone concentration response relationships for forest trees were also applied as published in LRTAP (2010).

4.2.4 Estimation of the reduction of C stored in the living biomass of trees

For the EMEP **meteorology** model runs, the tree biomass C reduction was calculated as the difference between the current C stored in trees and the C that would have been stored if ozone would not have had an impact on tree growth (called baseline scenario in the following), as modelled

by the DO₃SE model. In this study we could not estimate the impacts of ozone on soil C sequestration as no ozone flux-effect relationships are available for soil C pools.

Fractional area-weighted relative biomass (RBL_{grid}) as impacted by ozone for **all** species and cover types in each grid of the relevant modelling domain were estimated according to the species-specific relative biomass ($RB_{gridspec}$, fraction) predicted by the DO₃SE model and the land cover fraction of that specific species in the respective grid ($LCF_{gridspec}$, fraction) (eq. 4). In cases where a grid square extended over more than one climate region, the RB_{grid} was estimated using the appropriate climate specific parameterisation of the majority region.

$$RB_{grid} = \sum_{i=1}^n [RB_{gridspec} * LCF_{gridspec}] \quad [4]$$

Table 4.4 DO₃SE forest tree parameterisation for CCE and ME climate region species (LRTAP Convention, 2010). Parameterisations used for climate model runs are shaded in grey.

Parameter	Units	Continental Central Europe (CCE)		Mediterranean Europe (ME)	
		Beech and birch	Norway Spruce	Beech	Aleppo Pine
Land Use	EUNIS Class	Deciduous broadleaf forest	Coniferous forests	Deciduous Mediterranean broadleaf	Mediterranean needle leaf evergreen
Gmax	mmol O ₃ m ⁻² PLA s ⁻¹	150 (132-200)	125 (87-140)	145(100-183)	215
Fmin	(fraction)	0.13	0.16	0.02	0.15
Fphen(leaf_fphen)_a	(fraction)	0	0	0	1
Fphen(leaf_fphen)_b	(fraction)	(1)	(1)	(1)	1
Fphen(leaf_fphen)_c	(fraction)	1	1	1	0.4
Fphen(leaf_fphen)_d	(fraction)	(1)	(1)	(1)	1
Fphen(leaf_fphen)_e	(fraction)	0.4	0	0	1
Fphen(leaf_fphen)S	(days)	20	0	15	(0)
Fphen(leaf_fphen)E	(days)	20	0	20	(0)
Light_a		0.006	0.01	0.006	0.013
Tmin	°C	5	0	4	10
Topt	°C	16	14	21	27
Tmax	°C	33	35	37	38
VPDmax	kPa	1.0	0.5	1.0	1
VPDmin	kPa	3.1	3.0	4.0	3.2
LAlmin		0	12	0	1
LAlmax		5	12	5	2.5
LAls	(days)	15	n/a	-	100
LAla	(days)	20	n/a	-	166
Albedo	(fraction)	0.16	0.12	0.16	0.12
Height	m	25	20	20	10
Leaf dimension	cm	7	0.8	7	0.8

Table 4.4 (cont.) DO₃SE forest tree parameterisation for NE and ACE climate region species (replicated from LRTAP Convention, 2010).

Parameter	Units	Northern Europe (NE)		Atlantic Central Europe (ACE)		
		Norway Spruce	Birch and beech	Beech	Scots Pine	Temperate Oak
Land Use	EUNIS Class	Coniferous forests	Deciduous broadleaf forests	Deciduous broadleaf forest	Coniferous forests	Deciduous broadleaf forests
Gmax	mmol O ₃ m ⁻² PLA s ⁻¹	112 (111-118)	196 (180-211)	150 (100-180)	180 (171-188)	230 (177 – 325)
Fmin	(fraction)	0.1	0.1	0.1	0.1	0.06
Fphen(leaf_fphen)_a	(fraction)	0	0	0	0.8	0
Fphen(leaf_fphen)_b	(fraction)	(1)	(1)	(1)	(1)	(1)
Fphen(leaf_fphen)_c	(fraction)	1	1	1	1	1
Fphen(leaf_fphen)_d	(fraction)	(1)	(1)	(1)	(1)	(1)
Fphen(leaf_fphen)_e	(fraction)	0	0	0	0.8	0
Fphen(leaf_fphen)S	(days)	20	20	15	40	20
Fphen(leaf_fphen)E	(days)	30	30	20	40	30
Light_a		0.006	0.0042	0.006	0.006	0.003
Tmin	°C	0	5	0	0	0
Topt	°C	20	20	21	20	20
Tmax	°C	200	200	35	36	35
VPDmax	kPa	0.8	0.5	1.0	0.6	1
VPDmin	kPa	2.8	2.7	3.25	2.8	3.25
LAImin		0	0	0	4.5	0
LAImax		3	3	4	4.5	4
LAI _s	(days)	15	15	15	n/a	20
LAI _e	(days)	30	30	30	n/a	30
Albedo	(fraction)	0.16	0.16	0.16	0.12	0.16
Height	m	20	20	20	20	20
Leaf dimension	cm	0.8	5	7	0.8	5

Table 4.5 Dose-response relationships (LRTAP Convention, 2010) used for the calculation of flux-related biomass reductions. NE = Northern Europe, ACE = Atlantic Central Europe, CCE = Continental Central Europe, ME = Mediterranean Europe. BR = biomass reduction (%); RB = relative biomass (fraction).

	DO ₃ SE model runs				Reference
Input data	Climate data		Meteorological data		
Data source	RCA		EMEP		
Model reference	Kjellström et al., 2005		Simpson et al., 2003		
Parameterisation	Receptor	Dose-response relationship and parameter	Receptor	Dose-response relationship and parameter	Reference
Deciduous trees	Beech and birch (CCE)	RB = 1.00 – 0.011 * POD ₁ Whole tree biomass	Birch and beech (NE)	RB = 1.00 – 0.011 * POD ₁ Whole tree biomass	LRTAP Convention , 2010
			Beech (ACE)	RB = 1.00 – 0.011 * POD ₁ Whole tree biomass	Karlsson et al., 2007
			Temp. oak (ACE)	BR = 1.429 – 0.286 * POD _{1.6} Whole tree biomass reduction (%)	
			Beech and Birch (CCE)	RB = 1.00 – 0.011 * POD ₁ Whole tree biomass	
			Beech (ME)	RB = 1.00 – 0.011 * POD ₁ Whole tree biomass	
Coniferous trees	Norway spruce (CCE)	RB = 1.00 – 0.0024 * POD ₁ Whole tree biomass	Norway spruce (NE)	RB = 1.00 – 0.0024 * POD ₁ Whole tree biomass	LRTAP Convention, 2010
			Scots pine (ACE)	BR = 1.193 – 1.228 * POD _{1.6} Whole tree biomass reduction (%)	Karlsson et al., 2007
			Norway spruce (CCE)	RB = 1.00 – 0.0024 * POD ₁ Whole tree biomass	LRTAP Convention, 2010
			Aleppo pine (ME)	BR = 1.429 – 0.286 * POD _{1.6} Whole tree biomass reduction (%)	

The C stock held within each EMEP grid (C_{grid} , Mt C) in the year 2000 was estimated based on the Forestry Inventory dataset (Forest Europe, 2011; see Section 4.2.1) assuming an even distribution of C stock across the forested area of the country, the latter being delineated according to the JRC landcover map. The C stock held within each species and land cover class within each grid was similarly divided assuming an area weighted distribution.

The total whole tree biomass in the year 2000 per grid square in Mt C (B_{grid}) was then calculated according to equation 5, assuming a C fraction of the whole tree biomass of 0.47 as suggested by IPCC (2006):

$$B_{grid} = C_{grid} / 0.47 \quad [5]$$

The grid-specific baseline biomass (BB_{grid}) was subsequently calculated as follows:

$$BB_{grid} = B_{grid} / RB_{grid} \quad [6]$$

which enables the calculation of the grid-specific baseline C stock (BC_{grid} , Mt) as

$$BC_{grid} = BB_{grid} * 0.47 \quad [7]$$

The reduction of C in Mt C (LC_{grid}) due to ozone is the difference between BC_{grid} and C_{grid} :

$$LC_{grid} = BC_{grid} - C_{grid} \quad [8]$$

The C reduction percentage $LC_{grid\%}$ was finally calculated as follows:

$$LC_{grid\%} = LC_{grid} / BC_{grid} * 100 \quad [9]$$

For the RCA **climate** model runs, the tree biomass C reduction for the year 2000 was calculated using the same method as outlined above, except that the LRTAP Convention harmonised land cover map was applied. In addition, the tree biomass C reduction in 2040 ($LC_{grid2040}$) as compared to the reference year 2000 was calculated as follows:

$$LC_{grid2040} = C_{grid2000} - C_{grid2040} \quad [10]$$

where $C_{grid2040}$ is the C in living biomass per grid in 2040 as modelled by DO₃SE.

4.3 Risk of reduction of C storage in trees in Europe due to ambient ozone

4.3.1 Concentration- versus flux-based approach

Figure 4.3 shows the distribution of AOT40 and POD_γ – the latter is shown for the three different scenarios: standard, climate region specific (CRS) and deactivated soil moisture module (SMD_{off}) – as modelled by DO₃SE using EMEP meteorology data for the year 2000. There is a clear difference in spatial pattern between the AOT40 and POD_γ metric. While the AOT40 distribution is only driven by ozone concentration levels, with a clear increase from northern to southern Europe (see also **Table 4.6**), the POD_γ distribution is characterised by the prevalent meteorology as the main driver for stomatal functioning and hence ozone uptake. The standard and climate specific model runs lead to highest POD_γ values in both Atlantic and Continental Central Europe, where reasonably high ozone concentrations coincide with favourable ozone uptake conditions. Relatively lower POD_γ values in

Northern and Mediterranean Europe can be explained by lower ozone concentrations in the case of the former and a drier climate leading to a reduced uptake of ozone in the latter case. Applying climate-specific parameterisations for the flux-based approach lead to a relatively lower risk of ozone damage in the southern parts of northern Europe and the western part of Spain. If soil moisture effects are not taken into account, the more even distribution of POD_Y across Europe is mainly driven by temperature and air humidity effects, as well as the prevailing ozone concentration. Table 4.6 reveals that the European average POD_Y is indeed highest for the SMD_{off} scenario, whereas the average POD_Y values for the standard and CRS scenario are 19 and 28% lower, respectively.

Similar distribution trends can be seen for the AOT40 and POD_Y as modelled by DO_3SE using RCA climate and ozone for the year 2000 (Figure 4.4). Again, the AOT40 indicates the highest risk for ozone effects on forest ecosystems in Mediterranean Europe, whereas Central Europe represents the climate region of highest risk according to the POD_Y metric (see also Table 4.7). These typical patterns shown for both EMEP and RCA climate and pollution data are in-line with previous publications, showing a steeper gradient from northern to southern Europe for AOT40 than POD_Y .

4.3.2 Current versus future climate and atmospheric ozone concentrations in Europe

The RCA model runs enabled the comparison of current and future risks of ozone effects on agroforestry systems. The chosen meteorology and emission scenario for 2040, indicating a decrease in ozone concentrations during the coming decades, led to a much reduced risk in the future as compared to 2000, as shown by both reduced AOT40 and POD_Y values (**Figure 4.4**). However, the degree of decline in relative risk in the future as indicated by the two ozone exposure metrics is strikingly different, with the AOT40 suggesting only marginal risk residuals for the very south of Europe. In contrast, the POD_Y still indicates risks of ozone on European agroforestry systems, in particular in Atlantic Central Europe. This difference can be explained by the different ozone concentration thresholds above which the prevalent ozone concentration contributes to these two metrics: While ozone concentrations below 40 ppb do not contribute to the AOT40, they do to the POD_Y , down to approx. 20 ppb depending on the maximum stomatal conductance as defined in the species-specific parameterisation (LRTAP Convention, 2010). This indicates that in large areas of Europe ozone concentrations during the growing season of forests in 2040 are predicted to be between 30 and 40 ppb, hence the distribution difference shown in Figure 4.4.

4.3.3 Reduction of sequestered C in the living biomass of trees due to ozone

The predicted reduction in sequestered C in living biomass across Europe varies according to the i) climate and pollution data (EMEP vs. RCA), ii) ozone exposure metric (AOT40 vs. POD_Y) and iii) scenarios for the POD_Y approach (standard vs. CRS vs. SMD_{off}) as depicted in **Figures 4.5 to 4.8** and summarised in **Tables 4.6 and 4.7**.

In general, AOT40-based C reductions are predicted to be lower than POD_Y -based C reductions. This effect can especially be seen in northern and central Europe, but is smaller in southern Europe, where high AOT40 values indicate a similar risk of C reductions as compared to the flux-based approach. Average C reductions in EU-27+NO+CH in 2000 as compared to baseline C stocks (i.e. without any impacts of ozone on C sequestration) are in the range of 12% to 17% using the POD_Y approach and EMEP or RCA climate and pollution data, whereas the AOT40-based C reductions are approximately 8% and similar for both EMEP and RCA datasets. The AOT40-based reductions are similar to those calculated for temperate and boreal forests as described in Chapter 6 and the concentration-based (daily mean) reductions calculated by Wittig et al. (2009).

In absolute terms, the amount of C reduction in Eu-27+NO+CH in 2000 as compared to the baseline was between 1249 and 1929 Mt C using the POD_Y approach, whereas the AOT40 approach predicted reductions of approximately 800 Mt C. In all cases, the highest C reductions were reported for Continental Central Europe, followed by Northern Europe using the POD_Y and Mediterranean Europe using the AOT40 approach (**Figures 4.5 and 4.6**). The countries with consistently the highest absolute C reductions using the POD_Y approach included France, Germany, Poland, Romania, Italy and Sweden (Tables 4.6 and 4.7; see also Chapter 6).

When comparing flux-based absolute C reduction predictions for the year 2000 using EMEP or RCA input data, the latter predicts higher reductions, mainly due to higher POD_Y values resulting from more favourable climatic ozone uptake conditions. This effect is most pronounced for Continental Central and Mediterranean Europe.

In general, the different scenarios of the EMEP model runs reveal a ranking with regard to absolute C reduction estimates of standard < CRS < SMD_{off} (Table 4.6). However, the CRS scenario predictions are lower than the standard scenario predictions for Northern Europe, due to the fact that in the standard scenario one parameterisation for Norway spruce was applied for all conifers, whereas the CRS scenario includes a specific parameterisation for Scots pine, which is widely cultivated in northern Europe and much more sensitive to ozone than Norway spruce (Karlsson et al., 2007). As expected, the most prominent difference between the standard and SMD_{off} scenario can be found in Mediterranean Europe, where drought effects leading to a reduced stomatal uptake of ozone are most prevalent. Hence, when switching off the soil moisture deficit module of the DO_3SE model, higher C reductions are predicted in the Mediterranean area due to higher ozone concentrations as compared to the rest of Europe (as also supported by higher AOT40 values).

The relative C reductions are highest in Central Europe according to the flux-based approach using the standard scenario, followed by Mediterranean and Northern Europe (Tables 4.6 and 4.7). However, when the CRS scenario is used, Northern European countries show the highest relative C reductions, while even higher reductions are predicted for Mediterranean and Continental Central European countries when applying the SMD_{off} scenario. The AOT40 approach predicts the highest relative C reductions to be in Mediterranean Europe due to higher AOT40 values in that region.

The comparison of current (2000) and future (2040) climate and emission scenarios using RCA data clearly shows a decline of C reductions in the future as compared to baseline C stocks under the assumption of drastically reduced ozone concentrations and a stabilising climate represented here by the IIASA GEA-LOW-CLE scenario. This effect especially occurs using the AOT40 approach (C reduction decline of 83%, some countries such as for instance Finland and Sweden even show a C gain effect), but also to a lesser extent (31%) using the POD_Y approach (Table 4.7). The difference in these two approaches can be attributed to the differing contribution of ozone concentrations between approximately 20 and 40 ppb to these two ozone exposure metrics (see discussion above). The effect of a decline in C reductions in living forest biomass in 2040 using the POD_Y approach is most noticeable in Continental Central and Mediterranean Europe, which is mainly driven by the highest reduction in POD_Y in these regions due to most effective decreases in ozone concentrations.

4.4 Conclusions and recommendations

The aim of this study was to quantify the risk that current and future ozone concentrations might pose for the C sequestered in the living biomass of European forests, as predicted by flux-based deposition modelling approaches. In addition, comparisons with the concentration-based approach (AOT40) were made. The results indicate some consistent trends which can be summarised as follows:

- The flux-based approach indicates a high risk in Atlantic and Continental Central Europe, due to a combination of medium ozone concentrations and favourable climatic conditions for high ozone uptake in forest.
- The combination of climate region specific parameterisations (LRTAP Convention, 2010) and species-specific dose-response relationships (Karlsson et al., 2007; LRTAP Convention, 2010) give a more heterogeneous and hence potentially more realistic prediction of C reductions across Europe as compared to a standard parameterisation. The latter assumes that the deciduous and coniferous trees in Europe are adequately represented by parameterisations and dose-response relationships for beech and Norway spruce, respectively. The climate-region specific scenario revealed substantially higher C reductions as compared to the standard scenario.
- The deactivation of the SMD module of the DO₃SE model, which simulates drought-free stomatal ozone uptake conditions throughout Europe, led to an increase in C reduction, especially in the warmer and drier climates in Central and Mediterranean Europe. This stresses the importance of including soil water conditions when estimating C reduction.
- Under future (2040) lower emission and stabilising climate scenarios, the reduction of C storage in the living biomass of trees due to ozone is expected to decrease considerably as compared to current (2000) emission and climate conditions. This trend is mainly based on a reduction in ozone concentrations across Europe.
- Both climate and emission input datasets (EMEP vs. RCA) used in this study reveal similar trends with regard to C reduction estimates, with the RCA suggesting higher C reductions probably due to more favourable meteorological conditions for ozone uptake.
- The concentration-based approach (AOT40) predicts substantially lower C reductions compared to the flux-based approach (POD_y), especially in Northern and Continental Central Europe. The concentration-based approach indicates a high risk of ozone impacts on forests in the Mediterranean areas.

It should be noted that while the spatial patterns and temporal trends indicated above can be postulated with a considerable degree of certainty, the absolute figures of C reductions given in this report have to be interpreted very carefully. It should be remembered that these are for effects on living tree biomass only, and do not take into account any effect on soil C processes, including any direct or indirect ozone effects on below-ground processes that affect the rate of C turnover in the soil (see Chapter 2 and 5). We only highlight here the potential effects of ozone on C sequestration in the living biomass of forests in Europe. Several necessary assumptions for this modelling study will have contributed to the uncertainty associated with the results. These assumptions include a homogenous distribution of forest biomass across EMEP and RCA grids and a representation of European forest trees by only two to seven species (depending on the scenario used) in terms of model parameterisation and dose-response relationships. Furthermore, any future changes in forest management and land use currently occupied by forests have not been taken into account in the estimations of C reduction for 2040. For example, results in Chapter 6 highlight the importance of the gap between forest growth and harvest rate for C sequestration. Despite these limitations of the study, we are confident that the present report will be helpful for the ongoing discussion of ozone effects on C sequestration at a regional scale. The dependence of these effects on the prevailing physical and pollution climate have been clearly demonstrated and might hence be of use for the development of policies targeting the sustainability of C sequestration in European forests. Global climate change modelling should incorporate the impacts of ozone on vegetation to more accurately predict the impacts of the future climate on C sequestration.

Table 4.6 Forest C reductions in living biomass (grid average and total in Mt/C and % as compared to baseline), grid average POD_Y and grid average AOT40 for the year 2000, using the standard (POD_1), climate region specific (mixture of POD_1 and $POD_{1.6}$) and deactivated soil water module parameterisation (POD_1) in DO₃SE, and EMEP ozone concentrations and meteorology. For the standard parameterisation, C reductions in living biomass (grid average and total in Mt/C and % as compared to baseline) as calculated using AOT40-response relationships are also shown. Standard deviations in brackets. NE = Northern Europe, ACE = Atlantic Central Europe, CCE = Continental Central Europe, ME = Mediterranean Europe.

Climate region	Average $POD_{1/1.6}$	POD_Y -based total C biomass reduction	POD_Y -based average C biomass reduction	POD_Y -based average relative C biomass reduction	Average AOT40	AOT40-based total C biomass reduction	AOT40-based average C biomass reduction	AOT40-based average relative C biomass reduction
	mmol m ⁻²	Mt C	Mt C	%	ppm h	Mt C	Mt C	%
Standard								
NE	21.1 (6.7)	255.6	0.4 (0.3)	8.5 (4.1)	6.1 (3.7)	29.7	0.05 (0.10)	1.3 (1.9)
ACE	33.4 (5.7)	40.3	0.2 (0.4)	14.3 (5.2)	9.7 (3.8)	10.5	0.05 (0.17)	2.8 (2.5)
CCE	24.9 (7.0)	734.9	0.8 (0.6)	14.1 (5.9)	24.6 (5.9)	499.2	0.57 (0.45)	10.3 (5.3)
ME	19.3 (8.0)	218.2	0.3 (0.6)	11.9 (5.6)	28.7 (8.3)	252.5	0.40 (0.63)	14.2 (7.5)
Total/Average	23.2 (8.2)	1249.0	0.5 (0.6)	12.0 (5.8)	19.2 (11.2)	791.8	0.33 (0.49)	8.2 (7.3)
Climate region specific								
NE	18.1 (6.4)	553.1	0.9 (0.7)	14.5 (6.6)	6.5 (3.8)	-	-	-
ACE	34.0 (5.6)	44.3	0.2 (0.5)	15.0 (7.6)	9.7 (3.8)	-	-	-
CCE	24.8 (7.3)	876.5	1.0 (0.9)	14.8 (7.1)	24.8 (5.9)	-	-	-
ME	12.0 (7.0)	215.4	0.3 (0.6)	10.7 (7.7)	29.2 (8.4)	-	-	-
Total/Average	20.5 (9.6)	1689.3	0.7 (0.8)	13.7 (7.4)	19.5 (11.3)	-	-	-
Soil moisture module deactivated								
NE	21.4 (7.0)	267.5	0.4 (0.4)	8.8 (4.8)	6.1 (3.6)	-	-	-
ACE	34.1 (5.0)	43.1	0.2 (0.4)	15.1 (5.7)	9.6 (3.7)	-	-	-
CCE	30.3 (4.2)	1034.4	1.2 (0.8)	19.7 (8.6)	23.9 (5.6)	-	-	-
ME	31.7 (8.6)	437.4	0.7 (1.1)	23.4 (10.6)	27.2 (8.0)	-	-	-
Total/Average	28.6 (7.9)	1782.3	0.7 (0.9)	17.3 (9.9)	18.5 (10.7)	-	-	-

Table 4.7 Forest C reductions in living biomass (grid average and total in Mt/C and % as compared to baseline), grid average POD₁ and grid average AOT40 for the year 2000 and 2040, using the generic parameterisation in DO₃SE, and RCA climate and meteorology. For the generic parameterisation, C reductions in living biomass (grid average and total in Mt/C and % as compared to baseline) as calculated using AOT40-response relationships are also shown. Standard deviations in brackets. NE = Northern Europe, ACE = Atlantic Central Europe, CCE = Continental Central Europe, ME = Mediterranean Europe.

Climate region	Average POD ₁	POD ₁ -based total C biomass reduction	POD ₁ -based average C biomass reduction	POD ₁ -based average relative C biomass reduction	Average AOT40	AOT40-based total C biomass reduction	AOT40-based average C biomass reduction	AOT40-based average relative C biomass reduction
	mmol m ⁻²	Mt C	Mt C	%	ppm h	Mt C	Mt C	%
2000								
NE	22.5 (8.5)	316.7	0.4 (0.4)	10.4 (5.2)	7.3 (4.2)	41.8	0.1 (0.1)	2.1 (2.3)
ACE	38.1 (4.5)	60.6	0.3 (0.5)	21.2 (9.0)	12.5 (3.8)	13.2	0.1 (0.2)	5.0 (3.8)
CCE	32.9 (7.4)	1246.9	1.5 (1.2)	21.4 (8.1)	21.7 (4.3)	482.5	0.6 (0.5)	10.4 (5.5)
ME	24.6 (7.7)	304.8	0.5 (0.8)	14.7 (5.8)	29.1 (6.7)	253.6	0.4 (0.6)	14.1 (7.6)
Total/Average	28.0 (9.4)	1929.1	0.8 (1.0)	16.2 (8.3)	18.6 (10.0)	791.0	0.3 (0.5)	8.4 (7.2)
2040								
NE	20.4 (7.1)	270.9	0.4 (0.3)	9.3 (4.3)	2.4 (1.4)	-0.9	0.00 (0.02)	-0.3 (0.9)
ACE	36.1 (4.2)	52.9	0.2 (0.4)	19.8 (8.3)	6.4 (1.6)	4.4	0.02 (0.04)	2.3 (2.0)
CCE	25.1 (7.5)	821.4	1.0 (0.8)	15.7 (6.4)	4.9 (2.2)	68.5	0.09 (0.09)	2.1 (1.9)
ME	18.1 (6.9)	184.2	0.3 (0.5)	10.2 (3.9)	9.4 (3.3)	57.1	0.09 (0.12)	4.1 (2.5)
Total/Average	22.8 (8.6)	1329.5	0.5 (0.6)	12.6 (6.5)	5.5 (3.6)	129.1	0.05 (0.09)	2.1 (2.4)

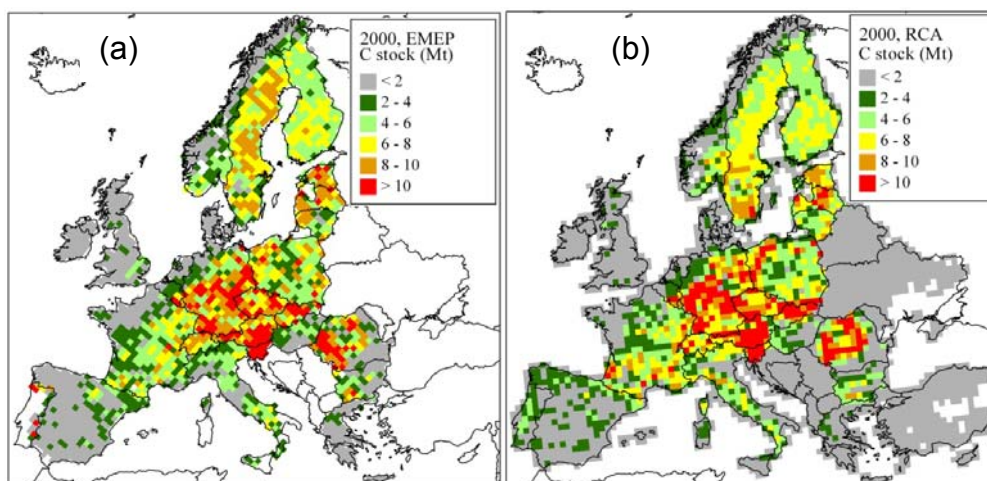


Figure 4.2 C stock in forests across Europe in 2000, based on forest inventory data (Forest Europe, 2011) and mapped on (a) the EMEP 50 x 50 km² grid and (b) the RCA 0.4° x 0.4° grid. For the EMEP grid the JRC land cover database was used as input data, for the RCA grid the LRTAP Convention harmonised land cover data was used.

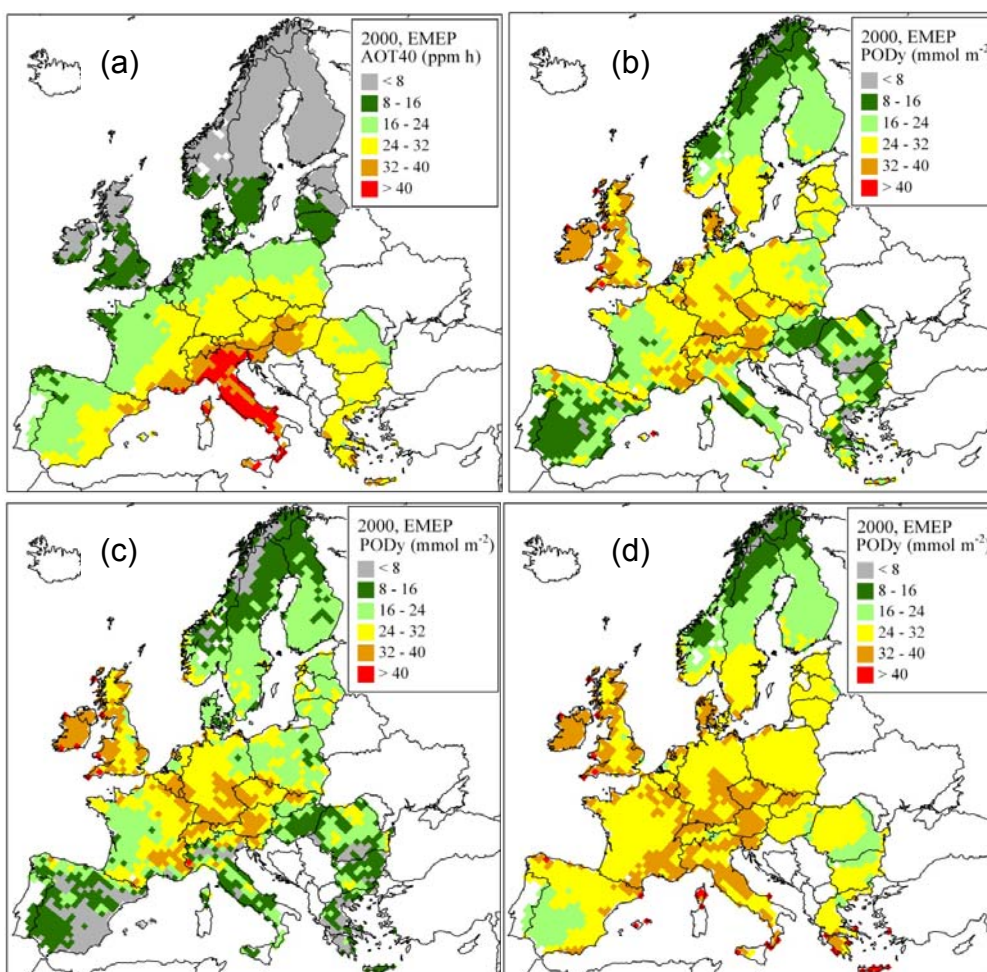


Figure 4.3 (a) AOT40 and (b) – (d) POD_y in 2000 calculated from EMEP input data and applying the following parameterisations in DO_3SE : (b) generic parameterisation ($Y = 1 \text{ nmol m}^{-2} \text{ PLA s}^{-1}$), (c) climate region specific parameterisation (Y is a mixture of 1 and 1.6 $\text{nmol m}^{-2} \text{ PLA s}^{-1}$), and (d) generic parameterisation with soil moisture module switched off (i.e. no soil water limitations).

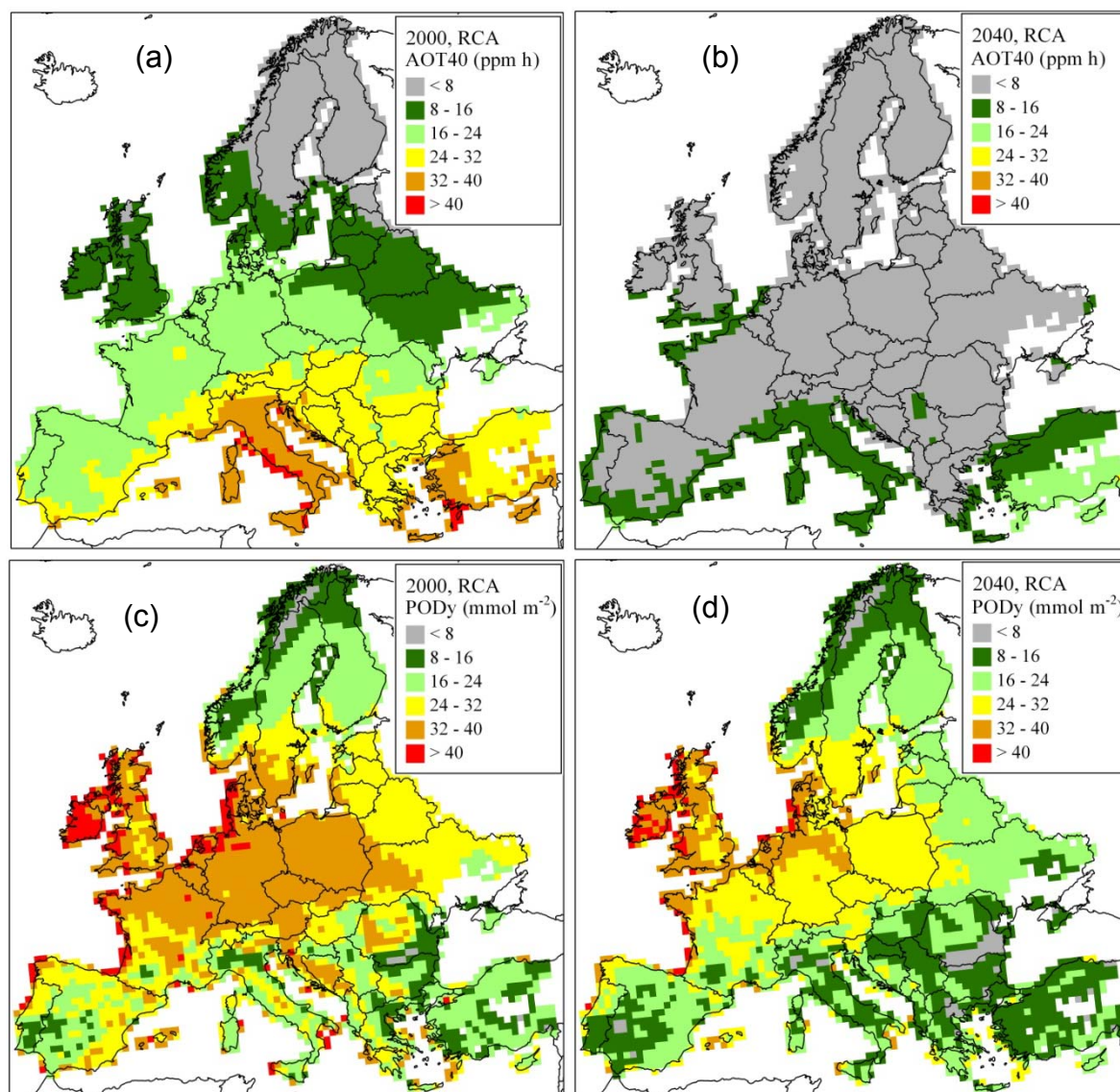


Figure 4.4 AOT40 in (a) 2000 and (b) 2040, and POD_y in (c) 2000 and (d) 2040, calculated from RCA input data and applying the generic parameterisation in DO₃SE ($Y = 1 \text{ nmol m}^{-2} \text{ PLA s}^{-1}$).

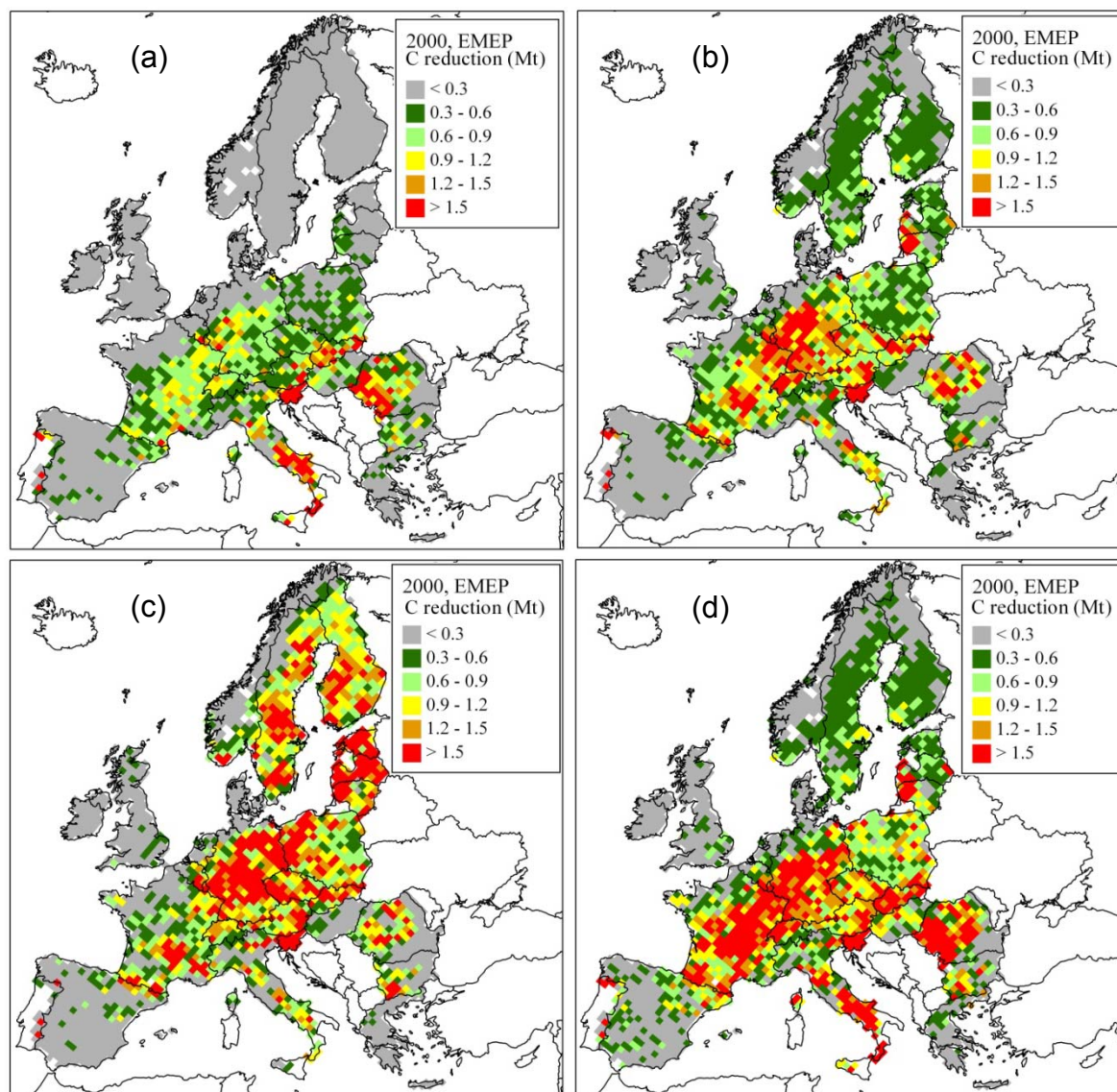


Figure 4.5 Absolute reduction (Mt C) in C storage in the living biomass of trees due to ozone in 2000, applying (a) AOT40 and (b) – (d) POD_y calculated from EMEP input data and applying the following parameterisations in DO_3SE : (b) generic parameterisation ($Y = 1 \text{ nmol m}^{-2} \text{ PLA s}^{-1}$), (c) climate region specific parameterisation (Y is a mixture of 1 and $1.6 \text{ nmol m}^{-2} \text{ PLA s}^{-1}$), and (d) generic parameterisation with soil moisture module switched off (i.e. no soil water limitations).

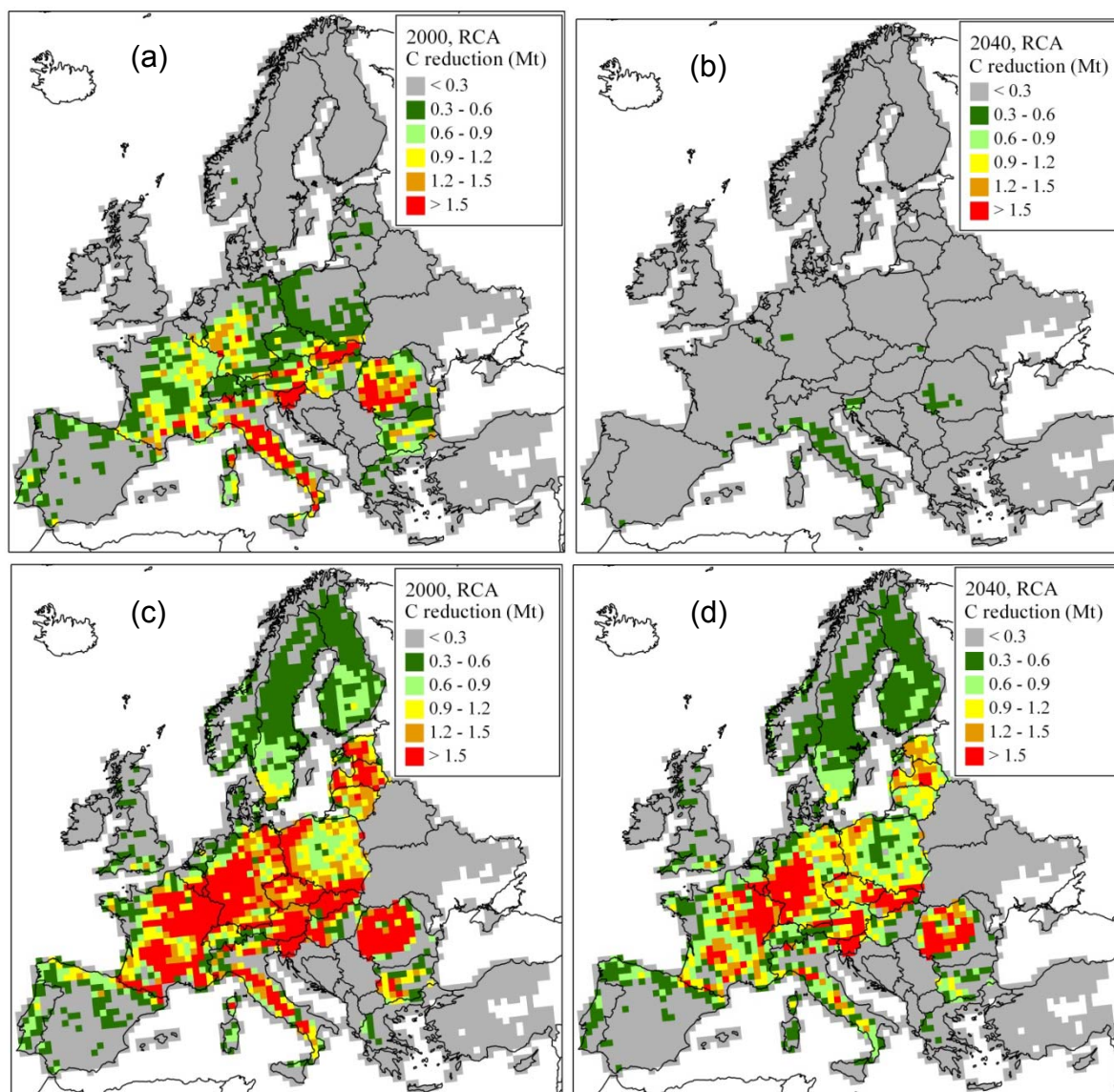


Figure 4.6 Absolute reduction (Mt C) in C storage in the living biomass of trees due to ozone applying AOT40 in (a) 2000 and (b) 2040, and POD_Y in (c) 2000 and (d) 2040, calculated from RCA input data and applying the generic parameterisation in DO_3SE ($Y = 1 \text{ nmol m}^{-2} \text{ PLA s}^{-1}$).

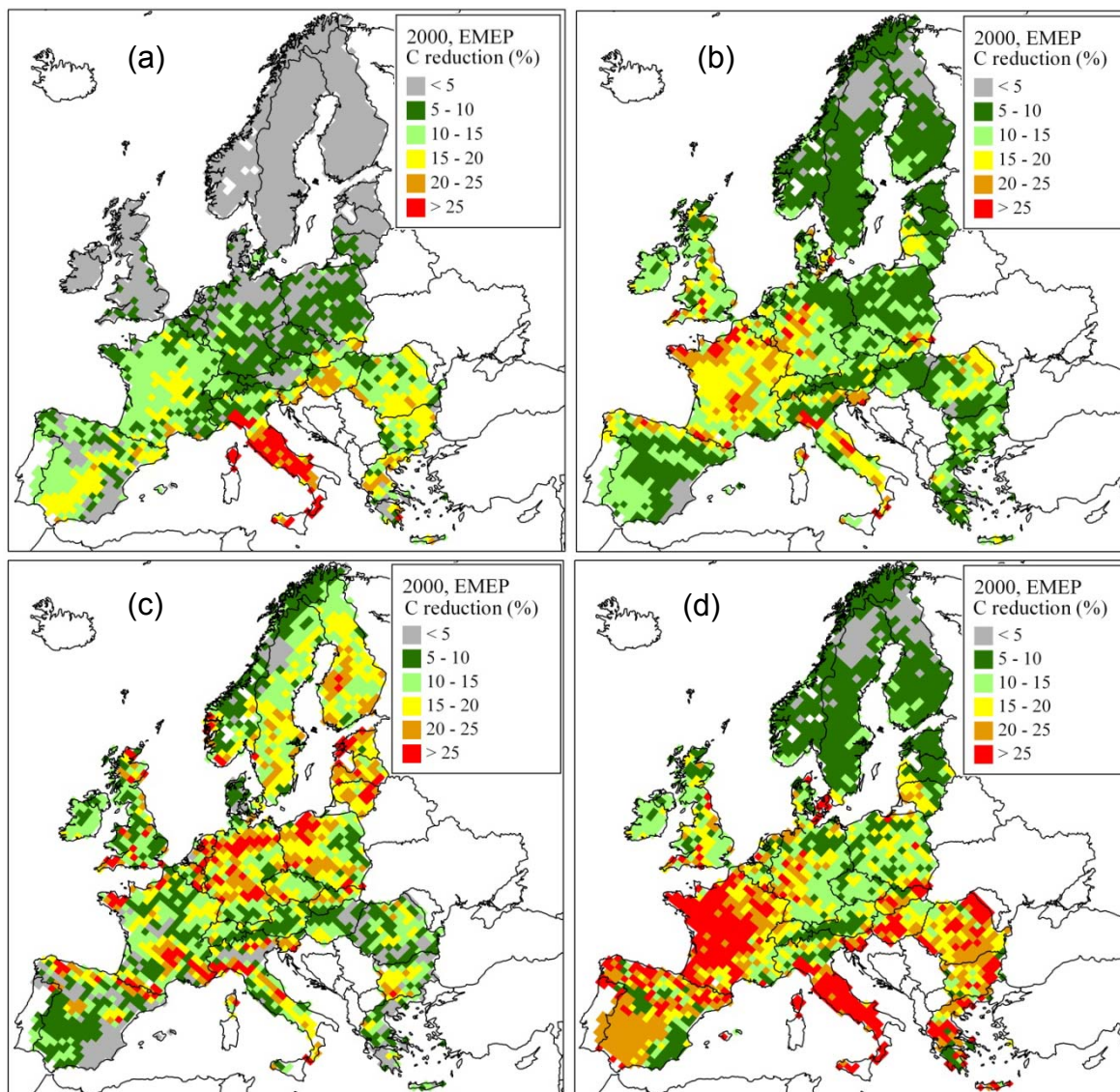


Figure 4.7 Relative reduction (%) in C storage in the living biomass of trees due to ozone in 2000, applying (a) AOT40 and (b) – (d) POD_Y calculated from EMEP input data and applying the following parameterisations in DO₃SE: (b) generic parameterisation (Y = 1 nmol m⁻² PLA s⁻¹), (c) climate region specific parameterisation (Y is a mixture of 1 and 1.6 nmol m⁻² PLA s⁻¹), and (d) generic parameterisation with soil moisture module switched off (i.e. no soil water limitations).

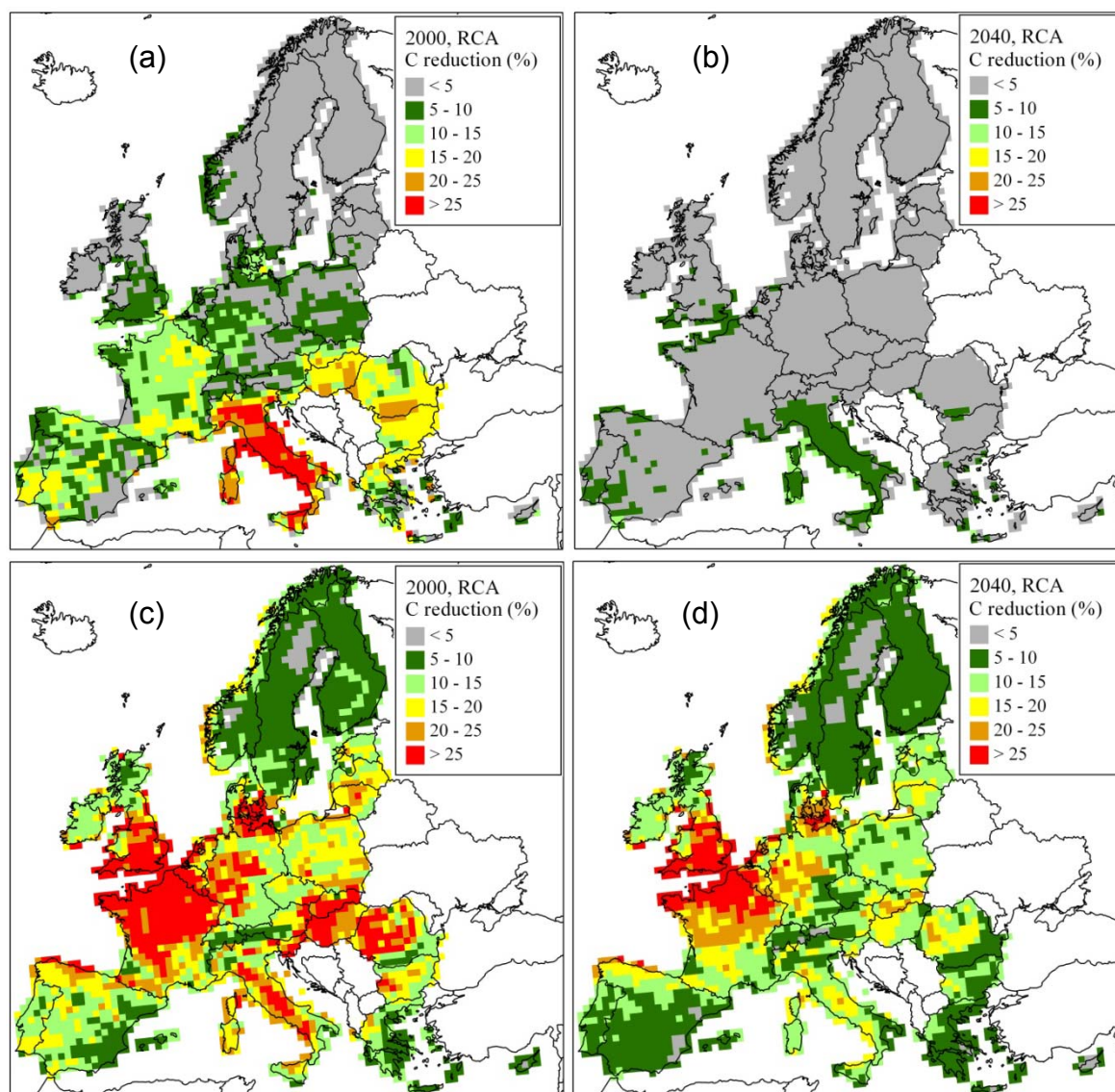


Figure 4.8 Relative reduction (%) in C storage in the living biomass of trees due to ozone applying AOT40 in (a) 2000 and (b) 2040, and POD_Y in (c) 2000 and (d) 2040, calculated from RCA input data and applying the generic parameterisation in DO_3SE ($Y = 1 \text{ nmol m}^{-2} \text{ PLA s}^{-1}$).

5 Ozone impacts on carbon storage in terrestrial ecosystems: a global perspective

Stephen Sitch, Stephen Arnold, Bill Collins, Chris Jones

5.1 Modelling ozone effects in JULES (Joint UK Land Environment Simulator)

In Chapter 4 we applied an empirical modelling approach using DO₃SE to assess the impacts of ozone on C storage in the living biomass of trees. In the current Chapter we followed a more mechanistic approach to modelling ozone effects on photosynthesis using JULES³ and the MOSES-TRIFFID⁴ land surface scheme (Essery et al., 2001; Sitch et al., 2007). In addition, the application of JULES allowed us to estimate globally the impacts of ozone on C storage in both vegetation and soil and the consequences for the global water cycle. The MOSES-TRIFFID land-surface scheme was modified, assuming a suppression of net leaf photosynthesis by ozone that varies proportionally to the ozone flux through the stomata above a specified threshold. The scheme includes an empirical relationship between stomatal conductance and photosynthesis (Cox et al., 1999) and through this mechanism the direct effect of ozone deposition on photosynthesis also leads to a reduction in stomatal conductance. As the ozone flux itself depends on the stomatal conductance, which in turn depends upon the net rate of photosynthesis, the model requires a consistent solution for the net photosynthesis, stomatal conductance and the ozone deposition flux (Sitch et al., 2007). Further details on how ozone effects on net photosynthesis and stomatal conductance are included in MOSES-TRIFFID are described in Sitch et al. (2007).

Recent studies, however, have shown that impacts of ozone are more complex than assumed previously and that ozone exposure might actually lead to an increase in stomatal conductance (e.g. Mills et al., 2009, Wilkinson and Davies, 2010). One of the aims of this case-study was to determine how uncoupling the stomatal response from net photosynthesis would affect the global C and water cycle (see Section 5.2) and how these results would compare to those published previously by Sitch et al. (2007). Therefore, in this case-study we applied slightly older flux-effect relationships than applied in Chapter 4. Data from ozone exposure experiments (Karlsson et al., 2004, Pleijel et al., 2004) were used to calibrate plant-ozone effects for the five Plant Functional Types (PFTs) described by JULES (see Sitch et al., 2007 for details on the calibration procedure). A ‘high’ and ‘lower’ parameterisation was chosen for each PFT to represent species sensitive (broadleaved trees) and less sensitive (conifers), respectively, to ozone effects (**Table 5.1**). The ‘low’ conifer parameterisation was assumed 3.8 times less sensitive than the high parameterisation (corresponding to the same ratio for the broadleaved trees). The flux threshold values used were 1.6 and 5 nmol m⁻² s⁻¹ for the woody and grass PFTs, respectively. Although a threshold of 5 implies a smaller ozone dose for grasses, the gradient of the flux-response function (parameter (a), see Table 6.1) is larger, and therefore grasses may become more sensitive to ozone exposure than trees at high ozone concentrations. For shrubs we assume the same plant-ozone sensitivity as broad-leaf trees. We prescribe some agricultural lands based on the HYDE croplands dataset (Klein Goldewijk, 2000), fixed at present-day coverage throughout the simulations, in which grasslands are assumed to be dominant.

³ Joint UK Land Environment Simulator; www.jchmr.org/jules/ (also for details on MOSES – TRIFFID)

⁴ Met Office Surface Exchange Scheme - Top-Down Representation of Interactive Foliage and Flora Including Dynamics

Table 5.1 Ozone exposure parameters: values for Broad-leaved Tree (BT) & Needle-leaved Tree (NT) calibrated to Karlsson et al. (2004; table 4). 'High' and 'Low' plant ozone sensitivity parameter 'a' calibrated against regressions for "Birch, beech" and "Oak", respectively. Parameters for C3 and C4 grass (C3, C4) were calibrated against data from Pleijel et al. (2004), with 'High' and 'Low' plant ozone sensitivity parameter 'a' calibrated against regressions for "Spring Wheat" and "Potato", respectively. Parameters 'a' for Shrub are calibrated against the same regressions as used for BT (Sitch et al., 2007).

	BT	NT	C3	C4	Shrub
Flux threshold ($\text{nmol m}^{-2} \text{s}^{-1}$)	1.6	1.6	5.0	5.0	1.6
'High' a ($\text{mmol}^{-1} \text{m}^{-2}$)	0.15	0.075	1.40	0.735	0.10
'Low' a ($\text{mmol}^{-1} \text{m}^{-2}$)	0.04	0.02	0.25	0.13	0.03

5.2 Projections of ozone effects on the global C and hydrological cycles

The UK Met Office's chemistry climate model has been used to generate monthly mean surface ozone concentrations for the present day (2000) and for the future (2050, with enactment of current pollution controls, CLE IIASA B2). JULES has been run with these ozone fields and observed climatology over the period 1901-2050 and results have been extracted for the periods 1901-2000 and 2000 – 2040. Results have been compared with the earlier findings of Sitch et al. (2007; see Ainsworth et al., 2012). The plant ozone damage implemented in JULES directly impacts plant photosynthesis, and thereby indirectly, leaf stomatal conductance. Leaf stomatal conductance regulates the intake of CO_2 for photosynthesis and the loss of water to the atmosphere, via transpiration. No direct effect of ozone on stomatal functioning is currently incorporated into JULES. With elevated near surface ozone levels, the model simulates decreased plant productivity, and as less CO_2 is required for photosynthesis, reduced stomatal conductance. Therefore, the plant assumed to be preserving water supplies. Indeed, in a modelling study using JULES, Huntingford et al. (2011) has compared the physiological effect of different greenhouse gases on two ecosystem services, productivity (a surrogate for food production), and runoff (a surrogate for freshwater availability). Ozone has been found to have large negative and positive impacts on the provision of productivity and freshwater, respectively.

Evidence of ozone effects on stomatal conductance are ambiguous as some studies suggest reduced stomatal functioning (Paoletti and Grulke, 2010) and an increase in stomatal conductance (Mills et al., 2009, Wilkinson and Davies, 2010) and thus increased plant water loss under elevated ozone concentrations, whereas a meta-analysis for trees by Wittig et al. (2007) suggests a decrease in stomatal conductance with ozone exposure. A mechanism to represent a direct ozone effect on stomatal conductance is not included in the current generation of models. As a test of the JULES model sensitivity to the stomatal conductance term we compared two model simulations: one (control) taken from Sitch et al. (2007), where the model was run from 1901-2100, with only near-surface ozone concentrations changing through time. Results were extracted for the periods 1901-2000 and 2000-2040. Here stomatal conductance is indirectly affected by ozone via its effects on photosynthesis, and thus CO_2 demand. In the absence of a formulation to describe the direct functional response of stomata to ozone, a new simulation (fixed stomata) was conducted where ozone affected photosynthesis as in Sitch et al. (2007), however, the stomatal conductance was not readjusted to take account of the modified leaf CO_2 demand. Broadly speaking this simulation represents the effect of elevated ozone on productivity, without its indirect effect on stomatal conductance. The same ozone fields were used as described in Sitch et al. (2007), i.e. future ozone

fields are consistent with the SRES A2 scenario (IPCC, 2001). This pessimistic future scenario was selected in order to give an upper bound estimate, appropriate with such a model sensitivity experiment. The results are shown in **Figure 5.1** and **Table 5.2**.

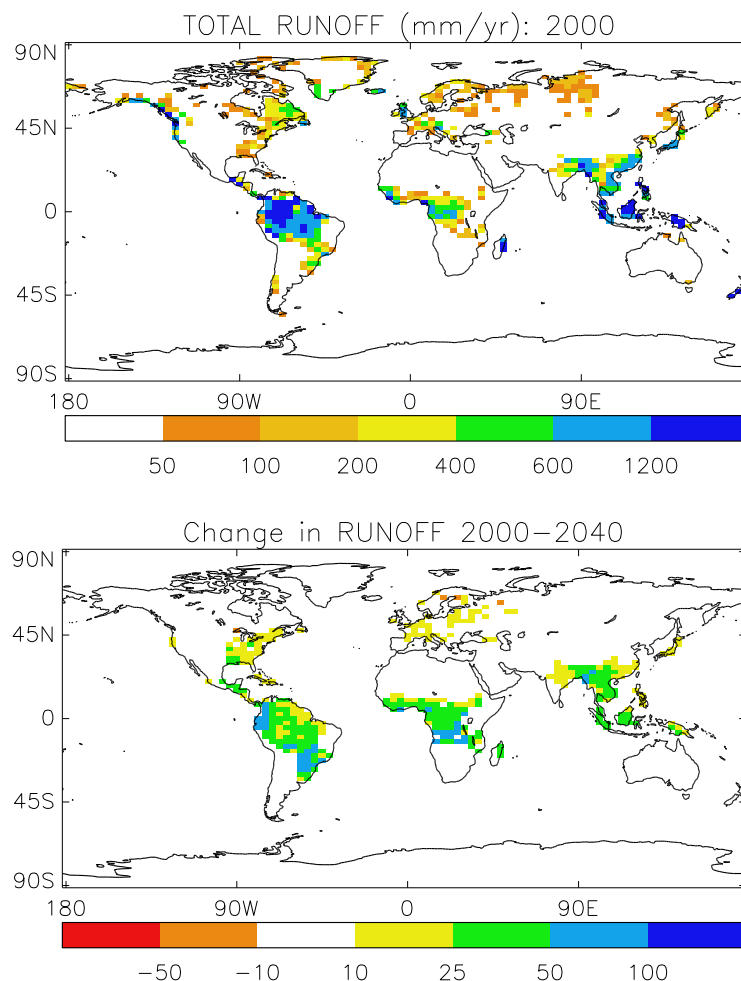


Figure 5.1 Effect of elevated ozone on runoff in 2000 (top) and the change in runoff between 2000 and 2040 (bottom) in the control simulation (see text for details).

As expected, results from the control run suggest a large indirect effect of ozone (via photosynthesis) on stomatal conductance and runoff. Stomatal conductance is simulated to decrease by between -8.7% and -13.3%, for the periods 1901-2000 and 1901-2040, respectively. Ozone concentrations are simulated to increase in the future under the SRES A2 scenario. These results compare favourably with observed reductions in stomatal conductance for trees of -13%, -10% and -6% when ambient vs charcoal filtered air (zero ozone), elevated vs charcoal filtered, and ambient vs elevated experiments respectively (Wittig et al., 2007). Unsurprisingly, stomatal conductance and river runoff changed little through time in the fixed stomata simulation, where the indirect effect of ozone on stomata via photosynthesis was switched off. However, despite the difference in stomatal conductance response between simulations, the differences in the response of the C cycle are only rather modest. The combined direct ozone effect on photosynthesis plus the indirect effects via stomatal conductance was simulated to reduce GPP by 15.4% with the direct effect alone reducing GPP by 17.9%. It can be concluded that in the absence of a direct effect of ozone on stomatal conductance, ozone vegetation impacts act to increase river runoff and freshwater availability substantially due to a reduced water loss from soil via transpiration from vegetation. However, such an increase might not occur if ozone has adverse impacts on stomatal functioning.

Table 5.2 Simulated future percentages changes (% Δ) in carbon (C) and water cycle (runoff) variables globally for three time periods: 1901-2040, 1901-2000 and 2000-2040. GPP = Gross Primary Productivity, Veg = vegetation, Gs = stomatal conductance (Scenario: SRES A2, IPCC, 2001). See text for further details.

1901-2040	% Δ GPP	% Δ VegC	% Δ SoilC	% Δ TotalC	% Δ Runoff	% Δ Gs
Control	-15.4	-10.9	-9.7	-10.0	12.6	-13.3
Fixed stomata	-17.9	-11.8	-10.5	-10.9	1.4	-1.6
2000-2040						
Control	-6.9	-5.0	-4.1	-4.4	4.5	-5.0
Fixed stomata	-8.1	-5.5	-4.6	-4.8	0.6	-0.5
1901-2000						
Control	-9.2	-6.2	-5.8	-5.9	7.7	-8.7
Fixed stomata	-10.7	-6.7	-6.2	-6.4	0.8	-1.1

5.3 Ozone effects on biomass across Europe

Ozone effects on C sequestration were estimated for Europe for the high ozone sensitivity plant parameterisation under the pessimistic SRES A2 and the more moderate CLE IIASA B2 scenarios. The combination of high ozone sensitivity and pessimistic ozone scenario indicate the upper-bound impact of ozone on European C stocks. The simulations suggest declines of almost 4% in vegetation C during the 20th century (**Table 5.3**). The JULES model projects further reduction in biomass of between 1.4 – 2% over the coming 40 years attributed to enhanced plant damage caused by ozone. Combined with concurrent losses in soils of 6.2%, European C storage is projected to decrease by up to 5.5% between 2000-2040. The further reduction in biomass of vegetation by 2040 is in contrast to the predicted future rise in C sequestration in the living biomass of forests (Chapter 4). This can be explained by the different scenarios applied, i.e. an increase and decrease in future ozone concentrations in this case study and the study described in Chapter 4, respectively.

Table 5.3 Simulated ozone effects on biomass across Europe (23°W-35°E, 37°N-72°N) for two time periods, 1901-2000 and 1901-2040 (scenarios SRES A2 and CLE B2, IPCC 2001).

2000-2040	% Δ VegC	% Δ SoilC	% Δ Total C
SRES A2	-2.0	-6.2	-5.5
CLE B2	-1.4	-4.1	-3.6
1901-2000			
SRES A2	-3.9	-10.2	-9.2
CLE B2	-3.7	-9.6	-8.7

5.4 Projections of ozone effects in 2003: impacts of the summer heat wave in Europe

Here we analysed ozone and drought interactions in Europe applying the climate specifically for the year 2003, which was a very dry year across the whole of Europe. The global 3-D TOMCAT chemical transport model (CTM) (Arnold et al., 2005; Chipperfield, 2006) was used to simulate hourly global tropospheric ozone for the year 2003. TOMCAT is forced using offline ERA-interim meteorological data from the European Centre for Medium Range Weather Forecasts (ECMWF), at a horizontal resolution of ca. $2.8^\circ \times 2.8^\circ$ with 31 hybrid sigma-pressure levels from the surface to 10 hPa. Sub-grid transport from convection (Stockwell and Chipperfield, 1999) and boundary layer turbulence (Holstag and Boville, 1993) is parameterised. The modelled tropospheric chemistry includes methane, NO_x , C2-C3 VOCs, isoprene photochemistry, wet and dry deposition (Giannakopoulos et al., 1999), and NO_x emissions from lightning (Stockwell et al., 1999). Anthropogenic emissions were prescribed using the IPCC AR5 2000 emissions set (Lamarque et al., 2010) along with biomass burning emissions climatology from the Global Fire Emissions Database version 2 for year 2003 (GFED2) described in Van der Werf et al. (2006). In addition biogenic emissions were prescribed from the POET (Precursors of Ozone and their Effects in the Troposphere) database, used as described in Emmons et al. (2010).

JULES was run over the period 1901-2003 with observed climatology from the Climate Research Unit (CRU), observed annual atmospheric CO_2 concentrations and modelled monthly ozone fields for the pre-industrial, present-day average and year 2003. The pre-industrial and present-day average fields were obtained from the STOCHEM model as used in Sitch et al. (2007) and year 2003 supplied by the TOMCAT chemistry transport model. For 20th century simulations with varying ozone concentrations, monthly ozone fields for the years between 1901 and 2000 were derived using interpolation between these pre-industrial and present-day average fields. For the years 2001 and 2002 ozone fields were taken at the present-day average, and TOMCAT monthly ozone fields were applied for year 2003. The following modelling experiments were conducted:

- S1) Changing CO_2 and ozone
- S2) Changing CO_2 , ozone and climate
- S3) Changing CO_2 , climate (pre-industrial ozone)

The ozone impact on biogeochemical cycles was diagnosed as the difference between the simulations S2 and S3. The climate impact was diagnosed as the difference between simulations S2 and S1. Results show a decline in annual gross primary productivity (GPP) in western Europe for 2003, primarily associated with the dry summer time conditions (**Figure 5.2**). Patterns are in broad agreement with Ciais et al. (2005). However, GPP is further decreased by the detrimental effects of elevated ozone on plant production (**Figure 5.2**). Elevated ozone was modelled to have an indirect effect on stomatal conductance through direct effects on the rate of photosynthesis. Ozone induced reductions in photosynthesis lead to reduced stomatal conductance and therefore increased runoff. In the absence of a direct effect of ozone on stomatal conductance, ozone acts to partially offset drought, i.e. ozone effects will act to conserve soil moisture and increase river runoff, as shown in **Figure 5.3** for year 2003. In summary:

- Large reductions in plant productivity are simulated under drought conditions. The net impact of ozone is to further reduce plant productivity under drought.
- In the absence of a direct effect of ozone on stomatal conductance, ozone acts to partially offset drought effects on vegetation.

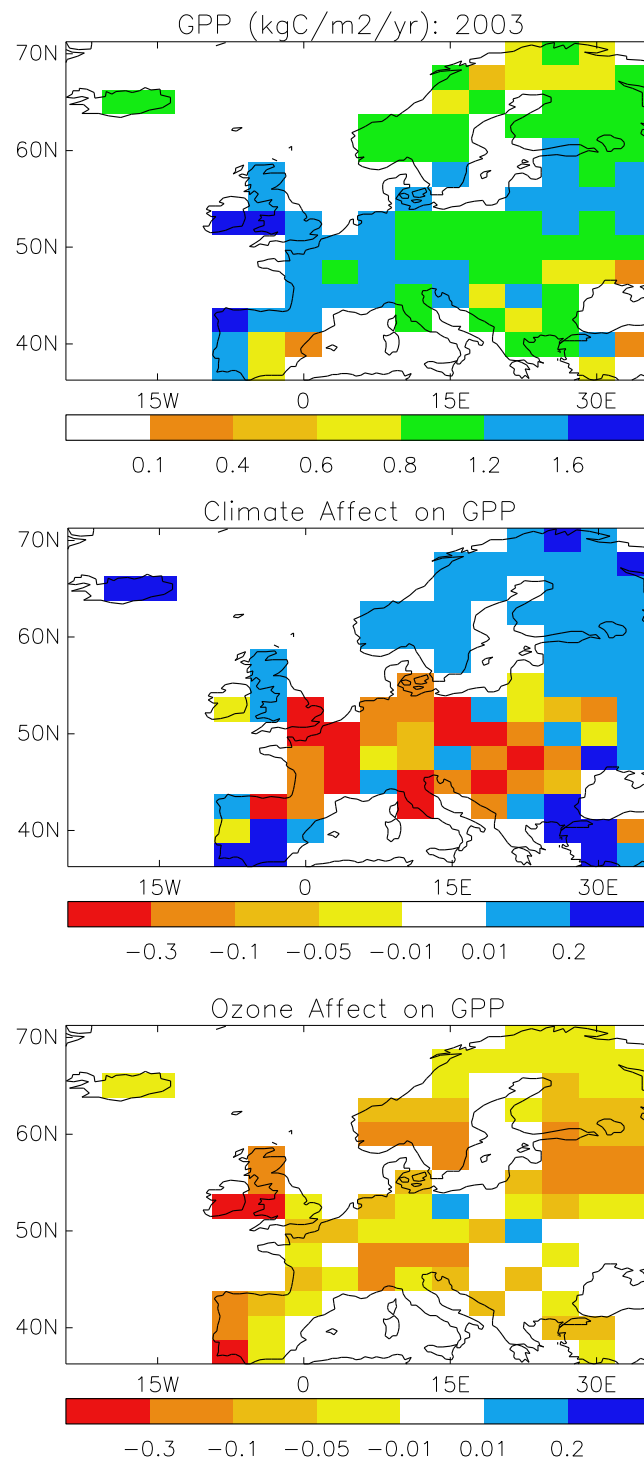


Figure 5.2 Top: Gross Primary Productivity (GPP) for the year 2003 in Europe in kg C m⁻² y⁻¹, Middle: Climate effect on GPP (calculated as S2 - S1, see text), Bottom: Ozone effect on GPP (calculated as S2 - S3, see text).

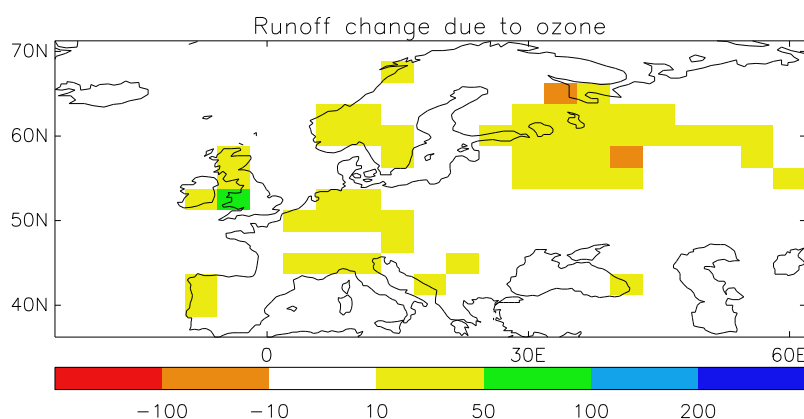


Figure 5.3 Changes in annual runoff (mm y^{-1}) in 2003 associated with ozone pollution.

5.5 Climate change, radiative forcing and novel metrics

Plant-ozone interactions have a large indirect effect on radiative forcing via the C cycle (Sitch et al., 2007). Using the same methodology as applied in Sitch et al. (2007), **Table 5.4** shows the indirect radiative forcing of ozone for two emission scenarios (SRES A2, IIASA CLE; IPCC, 2001) by 2040. Results suggest that by 2040 the more moderate Current Legislation Scenario implies a 25% lower estimate of radiative forcing due to indirect effects of ozone on the C cycle than the more pessimistic SRES A2.

Table 5.4 Indirect effect of ozone on radiative forcing (units: W m^{-2}) for two contrasting climate change scenarios (IPCC, 2001).

Ozone sensitivity vegetation		2000	2040
SRES A2	Low	0.226	0.376
	High	0.414	0.683
IIASA CLE	Low	0.214	0.288
	High	0.404	0.532

The climate effects of different trace gases and implications of regional emission reductions were investigated applying the novel Global Temperature Change Potentials (GTP; Shine et al., 2007) approach. In this way, the impact of both short- and long-lived trace gases on global temperature can be compared at a definite point in time. JULES was run with a one-year pulse change in ozone concentration consistent with a 20% change in individual ozone precursor emissions (NO_x : N oxides, CO: C monoxide, CH_4 : methane, BVOC: biogenic volatile organic compounds) for large regions (e.g. South Asia, North America, Europe). The response of the global land C cycle to this one year pulse was then followed (Collins et al., 2010). Annual GTP were calculated for greenhouse gas concentrations of CO_2 (representing the indirect ozone effect), ozone (the direct effect) and CH_4 . The sum gives the overall effect on climate warming over time. Findings show that for an increase in NO_x emissions, the longer time scale cooling associated with reductions in methane oppose the short-term warming associated with ozone and carbon dioxide, so NO_x emissions are warming in the short term, but cooling in the long term. BVOC, CO, and CH_4 all cause warming as emissions increase. GTPs are a strong function of emission location, owing to the different vegetation responses on different continents (Collins et al., 2010).

5.6 Conclusions

Applying ozone stomatal flux response relationships in JULES, the model predicted that the reduction in C stored in vegetation is 6.2% globally and almost 4% in Europe in 2000 compared to 1900, and is predicted to rise to 10.9% globally and ca. 5 to 6% in Europe by 2040 due to a predicted rise in atmospheric ozone concentrations in the future emission scenario applied. In JULES there is a large indirect effect of ozone (via photosynthesis) on stomatal conductance (reduced) and runoff (enhanced). However, switching off the indirect effect of ozone on stomata via photosynthesis hardly affected the response of the C cycle to ozone. It can be concluded that in the absence of a direct effect of ozone on stomatal conductance, ozone-vegetation impacts act to increase river runoff and freshwater availability substantially due to a reduced water loss from soil via transpiration from vegetation. However, such an increase might not occur if ozone has adverse impacts on stomatal functioning, reducing their responsiveness to environmental stimuli. Whereas the net impact of ozone in dry years is to further reduce plant productivity under drought condition, ozone acts to partially offset drought effects on vegetation.

6 Ozone impacts on carbon sequestration in northern and central European forests

Per Erik Karlsson

6.1 Introduction

The world's annual fossil CO₂ emissions (including emissions from cement production) are approximately 25000 Mt CO₂e (CO₂ equivalents), with emissions from land-use change (mainly tropical deforestation) contributing an additional 5000 Mt CO₂e (IPCC, 2007). The vegetation in temperate and boreal ecosystems sequesters in the order of 5000 Mt CO₂e annually and most of this goes into forests (Royal Society, 2001; Hyvönen et al., 2007). Intact tropical forests are estimated to sequester an additional 5000 Mt CO₂e annually (Trummer et al., 2009). A recent estimate of C sequestration by total global forests was ca. 14000 Mt CO₂e, excluding C storage in harvested wood products (Pan et al., 2011), of which biomass C constituted close to 80%. European forests contributed around 10% of the global C sequestration in forests. Actions to expand the area of boreal forests in order to mitigate climate change have been criticized, since this can change the local albedo, increase the absorbance of heat radiation and thus cause local warming (Bala et al., 2006). However, this should apply mainly to land-use change (afforestation, reforestation) and not to the same extent for maintaining high growth rates for already existing forested land. Regarding the ten northern and central European countries included in the present analysis, the highest rates of C sequestration occur in southern Sweden, Germany and Poland (Karjalainen et al. 2003; Note: The Baltic countries were not included in that study).

In this chapter the impacts of ground-level ozone on C sequestration in northern and central European forest ecosystem are assessed. The analysis includes temperate and boreal forest ecosystems in some major forest producing countries. There are large uncertainties regarding ozone impacts on mature forest ecosystems. Hence, the aim of this analysis was to make a first estimate of how ozone might negatively affect C sequestration and to be transparent about the input values used for the analysis.

6.2 Quantification methods

6.2.1 A description of northern European forests

Some FAO statistics on forests in ten northern European countries are provided in **Table 6.1**. The forested areas are largest in Germany, Finland, Norway and Sweden. This applies also for the growing stocks, although it also includes Poland. Regarding the total amount of C stored in the forests, the highest values are in Poland, Germany, Sweden and Finland. The distribution of the forest area between different types and age-classes is shown in **Table 6.2**. It was assumed that the age-class <10 years represented young stands, the age-class 11-60 years represented highly productive stands while the age-class >60 years represented aging forests with lower production. The fraction of highly productive coniferous forests was particularly high for Denmark and Sweden and it was low for Estonia. Instead Estonia had a relatively high fraction of productive broadleaved forests, together with Lithuania. Sweden and Finland had low fractions of productive broadleaved forests. The fraction of young forests was relatively similar between the countries. This applies also for the fraction of old coniferous forests except that Denmark had a low fraction. The fraction of forests characterized as mixed was generally low.

Table 6.1 Forested areas, growing stocks and C stocks for ten major, northern and central European countries. The four highest values for each parameter are given in bold; n.a.: data not available. Source: FAO statistics, State of the World Forests, 2009.

Country	Extent of forest		Growing stock			C in biomass	
	Forest area 2005 (1000 ha)	Annual change 2000- 2005 (%)	Per hectare (m ³ ha ⁻¹)	Total (Million m ³)	Com-mercial (%)	Tonnes ha ⁻¹	Total (Mt)
Czech Rep.	2648	0.1	278	736	97	123	326
Estonia	2284	0.4	196	447	94	73	167
Latvia	2941	0.4	204	599	85	79	231
Lithuania	2099	0.8	190	400	86	61	128
Poland	9192	0.3	203	1864	94	97	896
Germany	11076	n.a.	n.a.	n.a.	n.a.	118	1303
Denmark	500	0.6	153	77	76	52	26
Finland	22500	0.0	96	2158	84	36	816
Norway	9387	0.2	92	863	78	37	344
Sweden	27528	0.0	115	3155	77	43	1170
All	90155						

6.2.2 Mechanisms for C stock changes in productive forests

In general, forests that are actively managed sequester C at much higher rates than non-managed forests (Eriksson et al., 2007; Hyvönen et al., 2007; Nabuurs et al., 2008; Pingoud et al., 2010). Any measures that increase the productivity of temperate or boreal forest, such as e.g. fertilization or a more favourable climate, are likely to increase the forest C sequestration (Hyvönen et al., 2007; Eggers et al., 2008).

Table 6.2 Description of the distribution of forest areas regarding age-classes and forest types for ten major, northern European forested countries. Distributions are shown as percentage of total forest area in each country. Source: UNECE statistics, Age Structure of Even-aged Forest and Other Wooded Land by Availability for Wood Supply and Forest Type, Age Class, Country and Year; n.a.; data not available. Data for Germany was obtained from <http://www.bundeswaldinventur.de/enid/4d7ce5a78ed8275860f9cf240b3ac7e3,0/76.html>.

Age (yrs)	<=10			11-60			> 60		
	Coni-ferous	Broad-leaved	Mixed	Coni-ferous	Broad-leaved	Mixed	Coni-ferous	Broad-leaved	Mixed
Czech Rep.	6	1	2	26	6	6	39	8	6
Denmark	8	5	0	49	16	0	6	15	0
Estonia	1	4	1	16	28	14	20	6	10
Finland	8	1	1	31	4	7	40	2	5
Germany*	6	4	1	27	11	1	30	22	1
Latvia	4	4	2	18	21	6	28	11	7
Lithuania	4	3	2	20	27	9	20	8	7
Norway	1	3	3	19	10	12	33	11	7
Poland	n.a	n.a	n.a	n.a	n.a	n.a	n.a	n.a	n.a
Sweden	14	1	1	40	4	10	27	1	3

* The age classes available for German forests were <20, 20-60 and >60 years.

Furthermore, forest C sequestration rates change with stand age (Pregitzer and Euskirchen, 2004; Eriksson et al., 2007; Lindroth et al., 2009), with stands acting as a C source at young age, as a strong sink at medium age and as a weak sink or being C neutral at high age. Annual national values for changes in C stocks in total forest land are reported to the Climate Convention (UNFCCC). Positive values indicate emission to, negative values uptake from the atmosphere. The patterns for the rates of changes in the forest C stocks over time vary for the different countries depending on the methods applied in the National Forests Inventories (NFIs). Sweden probably has the world's most extensive NFI, with ca. 30000 observation plots that are revisited every 5th year, i.e. 6000 each year (Sweden, NIR 2011). For all countries that report differentiated C stock changes in the forest ecosystems, it can be seen that the major part of the C that is sequestered goes into the living biomass C stock. Dead biomass (including humus) and soil C also contribute, but these rates are slower with organic soils being sources for C to the atmosphere.

The most important aspect of forest management for C sequestration is the rate of harvests in relation to the forest growth rates (**Figure 6.1**), i.e. the higher the rates of harvests compared to growth, the lower C sequestration will be. This aspect has to be analyzed on the landscape level and/or over long time periods, since individual stands are regularly harvested.

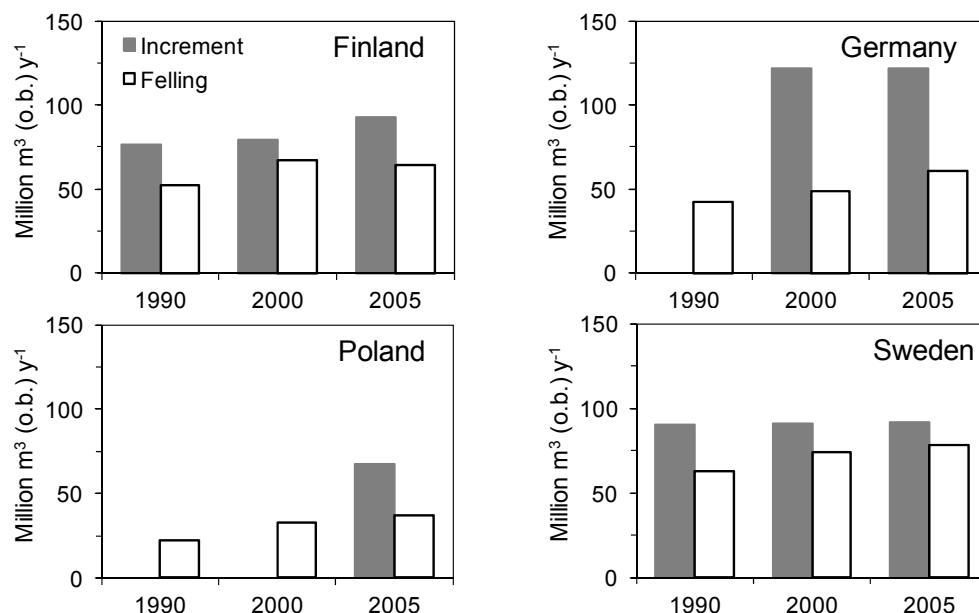


Figure 6.1 Annual national values for stem volume increment growth and annual fellings (Million m³ over bark (o.b.) per year) in Finland, Germany, Poland and Sweden. Values are from UNECE (UNECE, <http://www.unece.org/forests/welcome.html>), which in turn is based on FAO statistics.

6.2.3 Basic approach

For the assessment of today's ozone impacts on forest ecosystem C sequestration it is necessary to specify a number of definitions:

- **Baseline scenario:** in this study the baseline ozone scenario is defined as pre-industrial ozone levels, with concentration ranging from 10 - 15 ppb and no occurrence of ozone episodes, i.e. with AOT40 = 0.
- **Time horizon:** here we apply the general principle that is often applied in Life Cycle Analysis, i.e. that the assessment regards the current situation, as an average over a few years, and it does not

involve predictions for the future. However, in northern European countries, it has been estimated that forest production might increase substantially in a future climate (e.g. Poudel et al., 2011).

- Indicator: in the current study the focus is on the quantification of ozone impacts on the living biomass, the main C stock in forests.

6.2.4 Estimates of forest ozone exposure

As exposure – response relationships based on AOT40 are more commonly reported in the literature, this approach was applied in the current study. The nationwide values for AOT40 used for different countries in this study are shown in **Table 6.3**. However, it should be noted that the AOT40 approach might underestimate the risk of ozone impacts on vegetation in northern European countries in particular (e.g. Hayes et al., 2007b; Mills et al., 2011b; see also Chapter 4).

Table 6.3 Estimated nationwide mean values for annual, daylight AOT40 accumulated during the growing season for trees; the values are annual means for the time period 2000-2005. Source: EMEP model (David Simpson, 27-09-2011).

Country	AOT40 (ppm h)	Country	AOT40 (ppm h)
Czech Rep.	28	Latvia	10
Denmark	13	Lithuania	12
Estonia	7	Norway	4
Finland	3	Poland	21
Germany	24	Sweden	5

6.2.5 Derivation of ozone dose-response relationships

The response variable used for calculating the ozone effects in this study was the relative increment of either stem volume or total biomass, i.e. the increment during a period relative to the value at the start of the period. This period was usually the last year of the experiments. The impacts on the relative stem volume or biomass increments were related to the mean, annual daylight AOT40 during the entire experimental period. Unfortunately, many studies report only the percentage reduction of biomass caused by ozone at the end of the experiment and do not provide information on the biomass at the start of the experiment, so that impacts on growth rates cannot be calculated. For example, the results of the study by Wittig et al. (2009) could not be used in the current case study.

Ideally ozone impacts should be specified separately for coniferous and broadleaved tree species as well as separately for trees of different age classes. In this study ozone impacts were assessed differently for young trees before canopy closure (age <10 years), for productive age classes (age 10-60 years) and for old forests (age >60 years). Ozone exposure – response relationships for young coniferous trees were derived from Karlsson et al. (2005) (**Table 6.4**). The main reason to choose those results with Norway spruce (*Picea abies*) was that ozone impacts on relative growth rates could be determined. Furthermore, Skärby et al (2004) showed that the results from Karlsson et al. (2005) were comparable to the results from several other experiments with young Norway spruce from a large number of European studies.

The ozone exposure – response relationship for young broadleaved trees was also taken from Karlsson et al. (2005) (Table 6.4). This in turn was based on information on ozone impacts on total plant biomass of European silver birch (*Betula pendula*) saplings obtained from a two-year, open-top chamber experiment in Sweden (Karlsson et al., 2003). Results from this experiment were comparable with the results obtained from a number of open-air release experiments in Finland as well as with another open-top chamber experiment in Switzerland (Uddling et al., 2004).

The forest stem increment growth for the baseline scenario, i.e. the low ozone exposure, was calculated as:

$$y = h / (100 + (i * j) / 100)$$

where y = annual increment growth ($\text{m}^3 \text{y}^{-1}$), h = annual increment growth under current ozone exposure levels ($\text{m}^3 \text{y}^{-1}$), i = AOT40 (ppm h), j = the slope for the correlation between AOT40 and the per cent growth reduction ($\% (\text{ppm h})^{-1}$, negative values imply growth reductions).

Table 6.4 Ozone impacts on the annual stem volume increment growth rates of coniferous and broadleaved tree species separated into three different age classes, as related to the annual mean daylight AOT40, accumulated from 1 April to 30 September and expressed as ppm h. Ozone impact on the relative stem volume increment rates is expressed as % change.

Forest type	Age class ≤10 years	Age class 11-60 years	Age class >60 years
Conifers	-0.26 * AOT40	-0.26 * AOT40	-0.13 * AOT40
Broadleaved	-0.49 * AOT40	-0.49 * AOT40	-0.25 * AOT40

So far there are no indications that mature trees are less affected by elevated ozone concentrations compared to juvenile trees. On the contrary, Wittig et al. (2009) suggested that chamber studies on young trees might even underestimate ozone impacts compared to open-air field studies over longer periods. Therefore, it was assumed in this study that the same dose-response relationships apply for young trees and trees in the productive age. It is assumed however, that old trees (>60 years) have a lower ozone sensitivity due to overall reduced growth rates, here assumed to 50% of the growth rate of younger age classes (Table 6.4).

6.3 C sequestration in the living biomass of trees: impacts of ozone

The current annual, gross stem volume increment growth was estimated for the ten countries based on statistical information from UNECE, valid for the year 2005. This information is provided as total growth and felling values for each country. Hence, these values had to be distributed among different forest types and age classes, based on the information shown in Table 6.2, in combination with the assumption of different, relative area-based growth rates for different forest types and age classes. These assumed relative growth rates per area were derived based on information from the Swedish NFI for southern Sweden and were:

Conifers ≤10 years, 0.2; Conifers 11-60 years, 1.5, Conifers >60 year, 0.8;
Broadleaf ≤10 years, 0.4; Broadleaf 11-60 years, 1.3; Broadleaf >60 years, 0.8.

These adjustments were made so that the overall stem increment growth matched the values reported in the UNECE statistics. The estimated annual, stem volume increment growth under current ozone levels and pre-industrial ozone levels are also shown in **Table 6.5** as the values before and after the slash respectively.

Table 6.6 Estimated annual harvest rates (Million m³ over bark yr⁻¹) for different forest types and age-classes as well as for total forests.

Age (years)	≤10			11-60			>60			
Country	Coni-ferous	Broad-leaved	Mixed	Coni-ferous	Broad-leaved	Mixed	Coni-ferous	Broad-leaved	Mixed	Total forest
Czech Rep.	0.196	0.072	0.086	6.66	1.34	1.56	4.75	1.003	0.751	17.2
Denmark	0.031	0.039	0.000	1.35	0.390	0.000	0.07	0.211	0.000	1.84
Estonia	0.006	0.095	0.019	1.39	2.103	1.142	0.81	0.241	0.411	5.7
Finland	1.072	0.138	0.227	30.0	3.43	6.58	18.1	1.03	2.56	64.5
Germany	0.712	1.052	0.120	25.0	8.64	0.69	12.8	9.92	0.54	61.0
Latvia	0.086	0.171	0.051	2.96	3.078	0.88	2.25	0.956	0.577	11.3
Lithuania	0.062	0.079	0.036	2.12	2.579	0.896	1.04	0.434	0.373	7.24
Norway	0.033	0.144	0.091	3.2	1.45	1.81	2.60	0.948	0.549	11.1
Poland	0.370	0.281	0.185	12.7	8.07	3.99	7.8	2.20	1.77	37.2
Sweden	2.153	0.171	0.276	46.6	3.66	10.55	14.9	0.68	1.74	78.1
All										295

The total harvest rates obtained from the UNECE statistics for 2005 for each country was distributed across forest types and age-classes using identical relative factors as used to distribute current growth. The results are shown in **Table 6.6**. The net stem increment growth was calculated for each country, forest type and age-class based on the values provided in Tables 6.5 and 6.6. These yearly increment values were then converted to C stock changes as described in IPCC's "Good Practice Guidance for Land Use, Land-Use Change and Forestry" (Penman et al., 2003), somewhat modified as described by von Arnold et al. (2005):

$$\Delta C = Iv * BEF * D * CF$$

where: ΔC = C sequestration to tree living biomass (tonnes C ha⁻¹ y⁻¹);

Iv = yearly increment of timber volume (m³ ha⁻¹ y⁻¹);

D = density stem (tonnes dry weight m⁻³);

CF = "C fraction" of dry matter (tonnes tonnes⁻¹);

BEF = biomass expansion factor, converts between stem biomass and total living biomass including branches, leaves and roots.

The value of ΔC was then converted to CO₂-equivalents (CO₂e) by multiplying with 3.67.

The differences between the changes for the living biomass C stocks in the current ozone exposure and the pre-industrial ozone exposure scenario are shown in **Table 6.7**, in absolute values for the different forests types and age-classes as well as the total forests in each country and as percentage change for the total forests. The estimated percentage reduction in the change of the living biomass C stock across total forests in all ten countries was 10%. For different countries these values ranged between 2 - 32 %. The differences depended on the size of the gap between growth- and harvest rates, as discussed in the following section.

The calculated C stock changes of 56 Mt CO₂e yr⁻¹ in the living biomass in Finland, Norway and Sweden is in the same range as that presented by Pan et al. (2011) for the boreal forest C stock increase for the three countries since the Industrial Revolution, i.e. 77 Mt CO₂e yr⁻¹. Although the value calculated in the current study also included temperate forests, temperate forests cover a relatively small part of these countries. Hence, the assumptions made in this study were reasonable regarding forest types and age-class distributions, as well as the conversions between stem volume increments and C stock changes, at least for these three Nordic countries. Although percentage

changes in tree biomass at the end of an experiment due to ozone cannot directly be translated into changes in C stocks, the meta-analysis by Wittig et al. (2009) indicated a similar reduction in biomass (7%) as found in this study for C stocks (10%) when comparing current ambient levels with pre-industrial levels of ozone.

Table 6.7 Estimated reductions in annual C sequestration ($\text{Mt CO}_2\text{e yr}^{-1}$) due to current ozone exposure as compared to pre-industrial ozone levels for different forest types and age-classes as well as for total forests. Also presented is the percentage reduction due to ozone exposure, for the total forest in each country.

Age (years)	≤10			11-60			>60				
Country	Coni-ferous	Broad-leaved	Mixed	Coni-ferous	Broad-leaved	Mixed	Coni-ferous	Broad-leaved	Mixed	Total forest	Reduction (%)
Czech Rep.	0.02	0.01	0.01	0.65	0.26	0.23	0.22	0.09	0.05	1.55	32.0
Denmark	0.00	0.01	0.00	0.13	0.07	0.00	0.00	0.02	0.00	0.24	5.8
Estonia	0.00	0.01	0.00	0.05	0.14	0.06	0.01	0.01	0.01	0.28	4.5
Finland	0.01	0.00	0.00	0.32	0.07	0.10	0.10	0.01	0.02	0.64	2.2
Germany	0.10	0.29	0.03	3.49	2.42	0.14	0.86	1.30	0.05	8.69	12.3
Latvia	0.00	0.01	0.00	0.11	0.23	0.05	0.04	0.03	0.02	0.51	8.8
Lithuania	0.00	0.01	0.00	0.10	0.23	0.06	0.02	0.02	0.01	0.46	13.8
Norway	0.00	0.01	0.00	0.07	0.06	0.05	0.03	0.02	0.01	0.24	1.8
Poland	0.04	0.06	0.03	1.37	1.73	0.64	0.41	0.22	0.14	4.64	12.8
Sweden	0.04	0.01	0.01	0.77	0.12	0.25	0.12	0.01	0.02	1.34	8.6
All										18.6	9.8

6.4 Evaluation and conclusions

The following **key assumptions** and **uncertainties** are associated with the current study:

- Ozone impacts on forest ecosystem C stock changes were assessed only as direct impacts on growth rates, no indirect impacts such as reduced vitality etc. were included;
- Forest harvest rates were assumed not to be affected by the different ozone scenarios and total harvest rates were distributed among forest types and age-classed as related to growth rates in the same classes;
- It was assumed that AOT40 could be used as a relevant ozone exposure index across all countries independent of differences in climate;
- Ozone impacts on growth were assessed on the nation-wide scale, no distinctions were made for sub-national differences;
- Estimates of ozone impacts on growth rates were derived mainly from experimental studies on young trees. It must be emphasized that **knowledge about ozone impacts on mature trees under stand condition is to a large extent incomplete and further research is strongly needed.**

Despite the uncertainties associated with this study, it can be concluded that today's levels of ozone exposure in northern and central Europe have the potential to reduce the rate of increase in the forest living biomass C stocks in the order of 10%, as compared to pre-industrial ozone exposure levels. This value is of a similar order of magnitude as implicated by modeling studies (e.g. Sitch et al., 2007). The most important factor that determines the changes in the forest living biomass C stock is the gap between growth and harvest rates. If this gap is small, then a certain growth reduction caused by ozone will have a relatively large impact on the C stock change, if the gap is large, then the ozone

impact will be smaller on a percentage basis. This explains why the relative ozone impact is similar in Sweden and Germany, despite ozone exposure being much higher in Germany. The gap between growth and harvests has been much larger in Germany compared to Sweden, at least until 2005. The by far most important countries for C sequestration to the living biomass C stocks in northern and central Europe are Sweden, Finland, Poland and Germany.

The assessment made in this study did not include C stock changes in other parts of the forest ecosystems, besides the living biomass of trees. Dead biomass and soil C also contributes to the C stock increases, although at lower rates. It can be assumed that these processes are affected by reduced growth rates as well, so that the negative ozone impacts on forest C stock changes might be even larger compared to what was calculated in this study. In absolute values, the estimated impacts of ozone on C sequestration in the selected ten northern European countries was 19 Mt CO₂e yr⁻¹. In addition, the ozone induced growth reductions will also result in an economic loss for the forest owners, since they can sell less roundwood to the forest industry (Karlsson et al., 2005). The annual, economic loss for the Swedish forests owners has been estimated to be approximately 40 Million Euro (Karlsson et al., 2006).

There are few experimental exposure studies with large, mature trees under stand conditions. One example is the Kranzberg Forest experiment in southern Germany, where mature Norway spruce and European beech trees were exposed to twice ambient ozone concentrations for 8 years in an open-air release system. Fumigation of mature trees of Norway spruce and European beech to twice ambient ozone concentrations induced a shift in the resource allocation into height growth at the expense of diameter growth (Pretzsch et al., 2010). Annual diameter growth was reduced on average 11% across both species, but significantly reduced only during some years of the total 8-year experimental period. Both Norway spruce and European beech shifted their resource allocation under ozone fumigation to height growth at the expense of diameter growth. For Norway spruce, the increased height growth compensated for the reductions in growth at the stem basis, so that the whole stem production showed no losses. For beech, the increase in height growth was not enough to compensate for the reduced diameter growth, so there was a significant reduction in stem volume increment in beech due to elevated ozone. The results from the Kranzberg Forest experiment demonstrated that the growth patterns of both Norway spruce and beech were indeed affected by the twice ambient ozone fumigation, and this could be detected despite the low number of replication. However reductions in stem volume growth were significant only for beech. Similar results were found for European birch saplings after two years of exposure to elevated ozone concentrations in open-top chambers, where the ozone treatment increased both the shoot/root ratio as well as the stem height/diameter ratio (Karlsson et al., 2003).

7 Potential for impacts of ozone on C sequestration in agricultural and grassland systems

Gina Mills, Gerhard Soja, Matthias Volk

The focus of this report has been on quantifying impacts of ozone pollution on C sequestration in the biomass of forest trees. Within the ICP Vegetation we have shown that ozone also impacts on crops and grassland species (e.g. Mills et al., 2011a,b), with effects such as reductions in seed production and root growth all having the potential to change the C budget of these systems. Within this chapter, we provide a brief overview of the factors for consideration, but do not provide any estimates of the extent of effects.

7.1 Introduction

Quantifying the C budget of agricultural and grassland production systems is widely regarded as more challenging than for forested ecosystems (Moors et al., 2010). Complications arise from estimating the influence of crop rotations, proportion of crop harvested, proportion of crop remaining in the ground in non-tillage, conservation tillage or ploughed systems, C inputs by manure treatments, fertilizer and pesticide inputs, etc. (Smith, 2012). In part because of this complexity, the default requirement of the UNFCCC reporting guidelines consider crops to be C neutral (Moors et al, 2010). However, several recently published studies challenge this approach and are showing how C budgets can be estimated for croplands by, for example, averaging over four years to take into account crop rotations (e.g. Kutsch et al., 2010) and by taking into account the impacts of tillage on soil organic C content (e.g. Baker et al., 2007). These, and other studies acknowledge the impact of year-to-year variation in climate on C balance of agriculture crops, but do not include the potential influence of ozone pollution, even though a recent study by the ICP Vegetation predicted that ozone in EU27+CH and NO could be reducing wheat (and other crop yield) by up to 14%, with effects greatest in the main wheat growing areas of France, Germany and the UK (Mills and Harmens, 2011).

Figure 7.1 provides a schematic of the C balance of croplands (re-drawn from Smith et al., 2010, Fig. 1) and indicates where ozone pollution has the potential to impact. Ozone affects gross primary production (GPP) by reducing photosynthesis and increasing repair respiration, thereby decreasing the net primary production (NPP) of crops and increasing C losses to the atmosphere. For many crops, this results in less assimilate being partitioned to the harvestable fraction (e.g. seeds, tubers), reducing the biomass removed from the field. Impacts on soil organic matter and microbial biomass are more difficult to generalise, but it is well known that chronic exposure to ozone reduces root growth (e.g. Wittig et al., 2009) and can impact on C exudation (Chen et al., 2009). A further complication is the added application of fertilisers by farmers to compensate for reduced crop vigour and yield. A simulation study performed in the USA indicated that a 46.2% beneficial effect of N fertilisation on NPP was reduced to 39.8% by ozone pollution (Feltzer et al., 2004). These authors estimated that ozone pollution reduced C sequestration by ca. $9 \text{ g C m}^{-2} \text{ y}^{-1}$ across much of the crop growing areas of the mid-west.

A series of papers have recently been published showing the effects of crop species on the C budget of croplands. For example, Moors et al. (2010), excluding lateral inputs from manure etc. or C fixation outside of the cropping period, showed an average loss of -38 g C m^{-2} per cropping period based on 45 cropping periods and 17 sites across Europe. The variation in the study was very large at 251 g C m^{-2} per cropping period. For maize, a crop relatively sensitive to ozone Mills et al. (2007a) (**Table 7.1**), the average C release, calculated as Net Ecosystem Exchange minus the yield was -269 g C m^{-2} with a standard deviation of 208 g C m^{-2} ($N = 9$). The pattern for wheat, an ozone sensitive crop was

more variable with 7 of the site/year combinations showing a net gain of C, and 8 showing a net release, averaging at a net release of -54 g C m^{-2} (standard deviation of 256 g C m^{-2}). Ceschia et al. (2010) also included C inputs to croplands as organic fertilisers (manure, sugarbeet residue, lime), seeds or tubers, together with losses from harvest and burning to calculate the net ecosystem C budget (NECB). The NECB ranged from a net gain of -258 g C m^{-2} for rice in Spain to a loss of 645 g C m^{-2} for a combined fennel and maize crop in Italy. Ozone sensitive pea crops also had a high NECB of a loss of $400 \text{ g C m}^{-2} \text{ y}^{-1}$, with the short growing season resulting in long periods of bare soil with little/no photosynthetic C gain. It would be interesting to examine these different directions of effects for different crops and years in relation to the ozone climate at the sites.

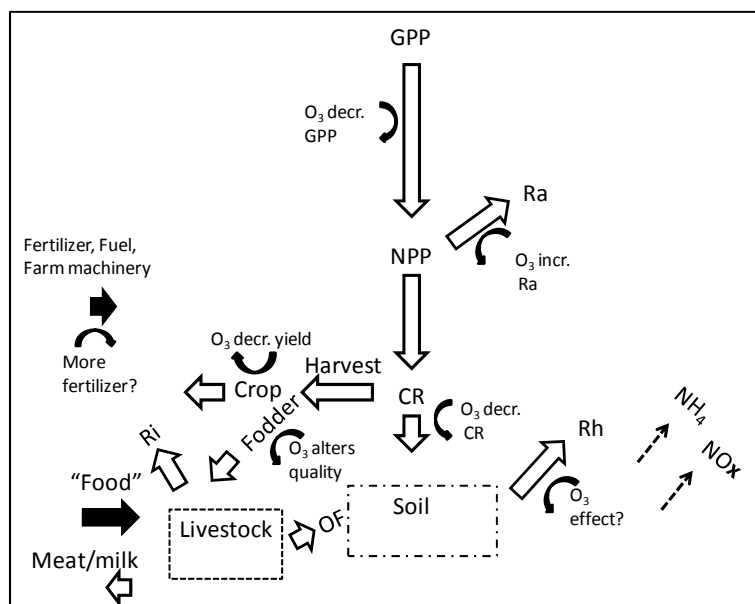


Figure 7.1 Schematic diagram, re-drawn from Smith et al. (2010), showing the C fluxes within croplands. The length of the block arrows gives an indication of the magnitude of the contribution, with $\text{GPP} = 100\%$, and filled arrows indicating imported C. Curved arrows show components of the C cycle that may be impacted by ozone pollution. Key: GPP: gross primary productivity; NPP: net primary Productivity; CR: crop residue; RI: respiration of the livestock; RA: autotrophic respiration from plants and mycorrhizae; Rh: heterotrophic respiration (microbial biomass and fauna); OF: organic fertilizer.

For grassland systems, the proportion of harvestable C (either removed by cutting or ingestion by herbivores) varies according to the intensity of production. In more productive grasslands the harvestable C proportion may reach up to 30% (Ammann et al., 2007), whilst in a dry year for a sub-alpine pasture, it can be as little as 2% of the annual C gain of GPP (Volk et al., 2011). But in a number of grassland studies (e.g. Cahill et al, 2009), including the 129 year analysis of the Rothamsted Park Grass experiment (Fornara et al., 2011), authors find no significant correlation between aboveground productivity and soil C stock size, but large difference in C turnover rates dependent on edaphic and climatic factors. The potential for ozone to impact on the C balance of sub-alpine grassland was investigated, for example, in the Alp Flix experiment (**Figure 7.2a**). The high ozone treatment had a significant, negative effect on GPP, but ecosystem respiration was reduced to the same degree, yielding an unchanged CO_2 balance (net ecosystem productivity; Volk et al. 2011).

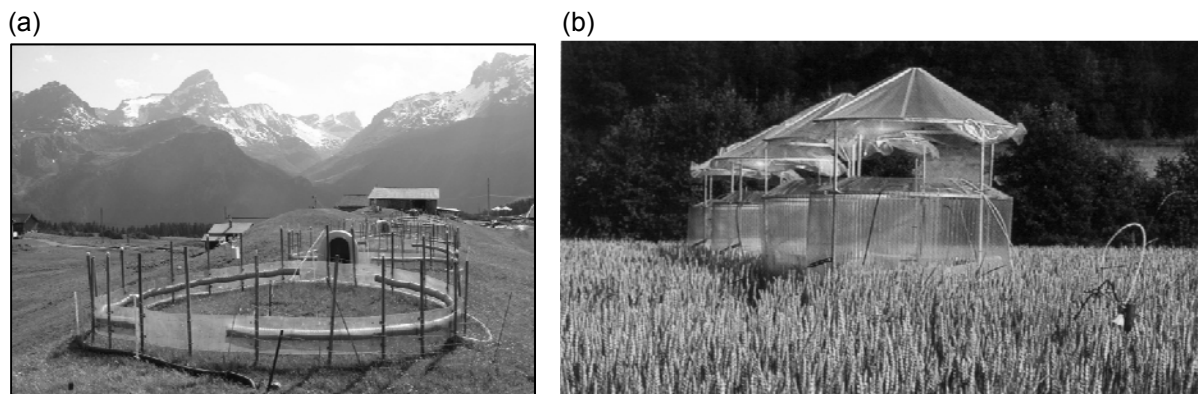


Figure 7.2 (a) Experiments investigating the combined effects of ozone and nitrogen pollution on sub-alpine grassland in Switzerland, Alp Flix site; photo from M. Volk. (b) Open-top chamber experiments in Sweden; photo from H. Pleijel.

7.2 Impacts of ozone on biomass (yield) removed from croplands

Many crops are sensitive to ozone within the range of concentrations experienced in Europe. Effects have been quantified by exposing crops growing in the field in open-top chambers (**Figure 7.2b**) which are placed over the crop as it emerges. The ozone concentration within the chamber is controlled by either filtration to remove ozone present in the air or by computer-controlled addition of ozone to either filtered or unfiltered ambient air. Microclimate within the open-top chamber is modified to a certain extent, but does fluctuate naturally with the climate. These types of experiments were conducted extensively in Europe and the USA in the 1980s and early 1990s, with fewer experiments with crops conducted since then.

Data from the open top chamber experiments were collated and analysed for crop sensitivity to ozone (Mills et al., 2007a), with an update presented in Mills and Harmens (2011) and reproduced in **Table 7.1**. Wheat, peas and beans, and soybean were found to be amongst the most sensitive group of crops, with maize, barley potato, oilseed rape and sugar beet being moderately sensitive. Mills and Harmens (2011) calculated that assuming soil moisture is not limiting to production, ozone impacts on wheat resulted in losses in production of 27 million tonnes of grain in 2000, falling to 16.5 million tonnes in 2020 under a current legislation scenario. The study showed that effects were likely to be greatest in parts of central Europe (e.g. Germany, France and Poland), as well as in some Mediterranean countries (e.g. Italy, Spain). Although not quantified here, it can be assumed therefore that for many crops, ozone is likely to be impacting on the amount of C removed from the field as harvestable yield. Given that the amount of C fixed is also likely to be lower, further studies are required to understand the consequences of this for the C cycle.

7.3 Potential impacts of ozone on C sequestration in the soil

Ozone can impact on the soil C pool by influencing root exudation of C for microbial activity or by altering the amount and/or quality of residual C matter entering the soil. We focus here on the second route as few studies have considered impacts on C exudation in crops.

Many earlier studies concluded that conventional ploughing in of stubble has been a primary cause of historical C loss from soils (e.g. Lal, 2004). Non-tillage farming has been advocated as a method of reducing C loss, and does indeed reduce C loss from the 0 - 15cm layers, however, more recent analysis has indicated that conventional tillage can increase C in the lower layers of the soil profile (Lal, 2007, Baker et al., 2007). For this study, a meta-analysis was conducted using Metawin software

to determine the potential for ozone to reduce C sequestration in soil by impacting on the quantity of residue left in non-tillage farming.

Data were collated from post-1997 published papers as indicated in **Table 7.2**. Selection criteria were that the data should be from field- or large container-based experiments involving ozone exposure over a growing season, with biomass fractioning between component parts of the plant reported. Exposure categories, based on 7 or 8h mean ozone concentration, were divided into 45 - 60 ppb, 61 - 80 ppb and 80 - 100 ppb, with results analysed as percentage change from the control treatment, usually charcoal filtered or non-filtered air. Crops included were soybean, rice, wheat, potato and sweet potato. Effects on the residual plant parts (i.e. those that were not harvested) were separated as reported into roots, shoots, stems and leaves. In potato studies, data for effects on tubers were excluded. Overall, using a Random Effects Model, ozone treatment caused a 20 - 30% decrease in residual biomass (**Figure 7.3a**), with effects being similar for all species except potato.

Table 7.1 Grouping of crops by sensitivity of yield to ozone. Values in brackets represent the percentage decrease in yield at 60 ppb ozone compared to that at 30 ppb, calculated from the regression equation (Mills and Harmens, 2011).

Sensitive	Moderately sensitive	Tolerant
Peas and beans (30)	Alfalfa (14)	Strawberry (1)
Sweet potato (28)	Water melon (14)	Oat (0)
Orange (27)	Tomato (13)	Broccoli (-5)
Onion (23)	Olive (13)	
Turnip (22)	Field mustard (12)	
Plum (22)	Sugar beet (11)	
Lettuce (19)	Oilseed rape (11)	
Wheat (18)	Maize (10)	
Soybean (18)	Rice (9)	
	Potato (9)	
	Barley (6)	
	Grape (5)	

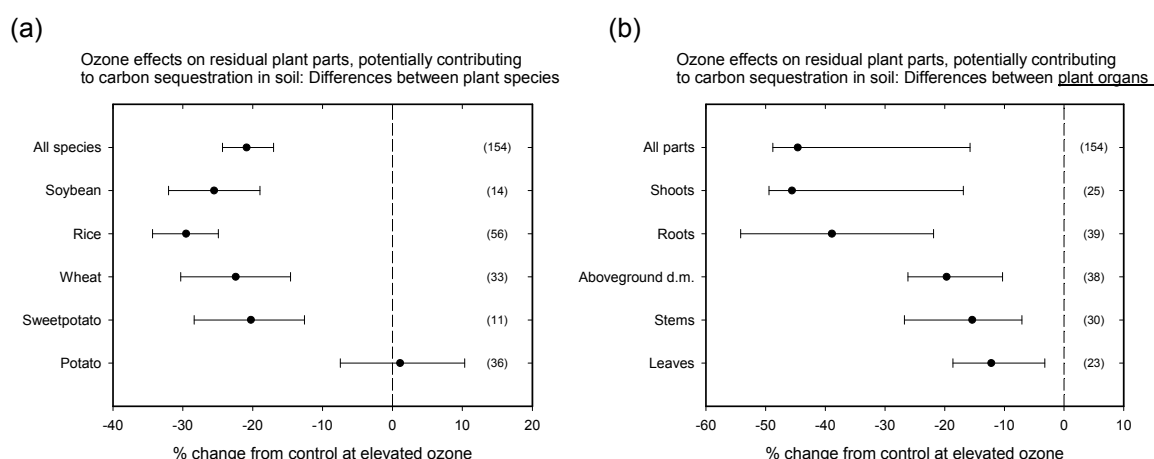


Figure 7.3 Meta-analysis of effects of ozone on non-harvestable parts of plants, split by (a) crop (Random Effects Model) and (b) plant part (Fixed Effects Model). Both non-filtered and C-filtered treatments were included as “control” treatments. The numbers in parentheses along the right axis are the degrees of freedom.

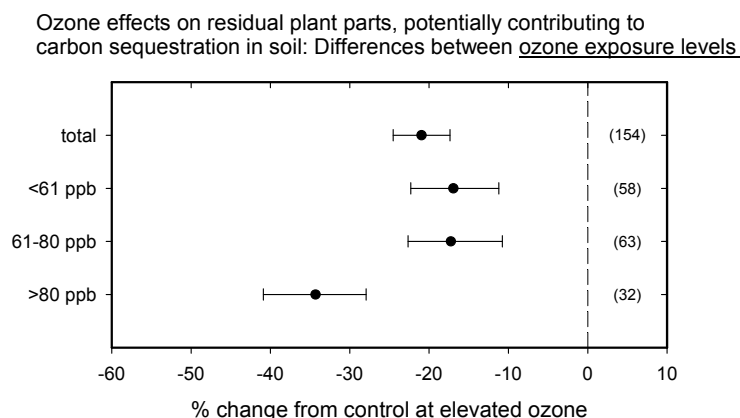


Figure 7.4 Meta-analysis of effects of ozone on residual parts of plants. Both non-filtered and C-filtered treatments were included as “control” treatments. The numbers in parentheses along the right axis are the degrees of freedom.

Analysis of effects on component parts independent of species, using a Fixed Effects Model, showed greater effects on roots than stems and leaves (**Figure 7.3b**). However, for those papers where effects on “shoots” were reported, the results showed a larger negative effect than indicated for papers where shoots were divided into stems and leaves. Across all species, the greatest negative effects were not surprisingly found for the 80 – 100 ppb treatment category (Random Effects Model). Of greater relevance to current and near future ozone concentrations, was the 18% reduction in crop residue predicted for concentrations in the range 45 – 60 ppb (**Figure 7.4**). Studies have shown that ozone treatment of wheat and rice impacts on the functioning of the soil microbial community, especially in the rhizosphere (Chen et al., 2009, 2010).

The above analysis has focussed on agricultural crop systems. For grasslands, we know that many component species are ozone sensitive (e.g. Hayes et al., 2007a), with effects such as reduced root biomass and enhanced or earlier senescence commonly reported. Studies to investigate the consequent impacts on below-ground C sequestration are ongoing.

Table 7.2 References used in the meta-analysis described in Section 7.3.

Crop	References used
Potato	Asensi-Fabado et al. 2010; Calvo et al. 2009; Craigon et al. 2002; Donnelly et al. 2001; Heagle et al. 2003; Köllner and Krause 2000; Lawson et al. 2001; Persson et al. 2003; Piikki at al. 2004
Rice	Akhtar et al. 2010b; Ariyaphanphitak et al. 2005; Frei et al. 2010b; Ishii et al. 2004; Maggs and Ashmore 1998; Reid and Fiscus,2008; Van et al. 2009
Soybean	Booker and Fiscus 2005; Booker et al. 2005; Jaoudé et al. 2008; Morgan et al. 2006; Robinson and Britz 2000; Singh and Agrawal 2011
Sweet potato	Keutgen et al. 2008
Wheat	Akhtar et al. 2010a; Ambasht and Agrawal 2004; Bender et al. 1999; Biswas et al., 2008; Biswas et al., 2009; Gelang et al. 2000; Hassan 2004; Heagle et al. 2000; Khan and Soja 2003; McCrady and Andersen 2000; McKee and Long 2001; Ojanperä et al. 1998; Pleijel et al. 1998; Pleijel et al. 2000; Cardoso-Vilhena and Barnes 2001; Chen et al., 2009; Gelang et al. 2001; Ainsworth 2008; Feng et al. 2008

7.4 Conclusions and further work

As discussed in the introduction, quantification of the dynamics of C sequestration within agricultural systems is complicated, with balances for crops often considered to be zero. In this chapter we have discussed how ozone has the potential to further complicate the C balance of crops by reducing CO₂ fixation, increasing repair respiration, and reduce biomass allocation to the harvestable crop, stubble and roots. There is also some evidence of ozone indirectly impacting on the microbial community within soils via effects on the crop. It is recommended that this area of research receives further study in order to fully understand the significance of these effects.

8 Conclusions and recommendations

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8.1 Ozone effects on C sequestration in the living biomass of trees

Apart from being the third most important greenhouse gas, ozone is also an important atmospheric pollutant and has adverse effects on human health and the environment. Hence, air pollution abatement policies with respect to ozone and its precursors are expected to have co-benefits for climate change. In addition, a recent modelling study has shown that the indirect impacts of ozone on the global warming potential via its negative impacts on vegetation are of similar magnitude as its direct impacts as a greenhouse gas (Sitch et al., 2007). The living biomass of trees is an important terrestrial C sink and many studies have shown that ozone reduces tree biomass production. Until this study was conducted, the quantification of impacts of ozone pollution on the living biomass of trees generally relied on the use of concentration-based ozone metrics such as AOT40 and 7h mean ozone concentration. In the current study, the ozone flux-based method was applied for the first time specifically to Europe to quantify the impacts of ozone on C storage in the living biomass of trees for the year 2000 and 2040. Ozone impacts on C storage in trees were estimated using the biologically relevant stomatal flux calculations, i.e. the Phytotoxic Ozone Dose above a threshold of $Y \text{ nmol m}^{-2} \text{ s}^{-1}$ (POD_Y ; LRTAP Convention, 2010; Mills et al., 2011a). The potential impacts of ozone on C sequestration in croplands and grassland was briefly discussed too, but the quantification of such impacts is currently associated with high uncertainties.

The impacts of ozone on C sequestration in terrestrial ecosystems in Europe in the future will depend on the interaction with and magnitude of other environmental and climate changes such as rising temperature, increased drought frequency, enhanced atmospheric CO_2 concentration and reduced nitrogen deposition. In contrast to the concentration-based approach, the stomatal flux-based approach for assessing the risk of impacts of ozone on vegetation has the ability to include such future environmental and climate changes (e.g. Emberson et al., 2000a) and therefore provides better estimates of the risk of ozone to vegetation in the future (e.g. Harmens et al., 2007). Ecosystems are inherently complex, and for any one aspect of functioning, there are multitudes of driving factors. Relatively few studies have investigated the interactive impacts of two or more drivers of long term change and at the field scale. The outcome of such studies often indicates complex interactions and non-linearity in responses. Recent reviews on field-based studies suggest that there is a general trend for the magnitude of response to decline with higher-order interactions, longer time periods and larger spatial scales (Long et al., 2005; Leuzinger et al., 2011). This suggests that on average, both positive and negative global change impacts on the biosphere might be dampened more than previously assumed.

Due to the lack of sufficient data from long-term, field-based, multi-driver studies, modelled predictions for the impacts of ozone and climate change on vegetation in the current and future climate are often based on the outcome of short-term, small scale, single driver studies in more or less controlled environmental conditions. For trees, exposure studies have often been conducted on juvenile trees. However, epidemiological studies (Braun et al., 2010) and field-based ozone exposure studies indicate that mature trees can be as sensitive to ozone as young trees or even more sensitive (see Wittig et al., 2009; Matyssek et al., 2010). As for experimental studies, modelling of future impacts of climate and environmental change should also be based on a multi-factorial approach. So far, the impacts of ozone on vegetation and feedbacks to the climate have hardly been considered in global climate modelling and other modelling studies, although some recent studies have now included ozone as a driver of change (e.g. Sitch et al., 2007; Collins et al., 2010; Huntingford et al., 2011).

The transboundary nature of ozone pollution requires international as well as national efforts to effectively reduce emissions of nitrogen oxides and volatile organic compounds and hence ozone impacts on C sequestration. Such emission reductions would have co-benefits for climate change and human health. Reducing ozone impacts on vegetation and health has been identified as a priority area in the Long-term strategy of the LRTAP Convention. To fully realise the benefit of international agreements there is an urgent need to develop effective policy interventions to reduce the threat to vegetation from ground level ozone. This would require an improved dialogue between policy makers, stakeholders and scientists which could then lend support to the procurement of scientific evidence appropriate for policy development, enhancing the development and modification of existing policies of relevance to ozone impacts on vegetation, and utilising opportunities for engagement across policy fields (e.g. air pollution and climate change issues) and across regions (e.g. UNECE and existing and developing air pollution policies in Asia).

8.2 Conclusions from this study

- Current levels of ambient ozone concentrations are reducing C sequestration in the living biomass of trees across Europe, with both episodic ozone peaks and background concentrations contributing to effects.
- Depending on ozone and source of meteorology/climate input data for the year 2000, parameterisation of the DO₃SE (Deposition of Ozone for Stomatal Exchange) model for generic deciduous and conifer trees predicted a reduction of C sequestration in the living biomass of trees by 12.0 (EMEP input data) to 16.2% (RCA input data) in comparison to pre-industrial ozone impacts.
- The climate-region specific parameterisation of DO₃SE for 2000 revealed higher C reductions (13.7%) due to ozone compared to the generic parameterisation for calculating POD_Y (12.0%).
- The deactivation of the soil moisture deficit module of the DO₃SE model, which simulates drought-free stomatal ozone uptake conditions throughout Europe, led to greater C reduction, especially in the warmer and drier climates in Central and Mediterranean Europe. Deactivation of the soil moisture deficit module predicted a reduction of C storage in the living biomass of trees of 17.3% across Europe compared to 12.0 % when the soil moisture deficit module was activated.
- By 2040, the reduction of C storage in the living biomass of trees due to ozone and climate change is predicted to have declined. Compared to pre-industrial C storage in the living biomass of trees, by 2040 the POD_Y-based C reduction is predicted to be 12.6% (compared to 16.2% in 2000).
- Whilst the spatial patterns and temporal trends indicated above can be postulated with a considerable degree of certainty, the absolute figures of C reductions given in this report have to be interpreted very carefully. It should be remembered that these are for effects on living tree biomass only, and do not take into account any effect on soil C processes, including any direct or indirect ozone effects on below-ground processes that affect the rate of C turnover in the soil.

- Conducted for comparison purposes, concentration-based analyses (AOT40) indicated that current levels of ambient ozone reduce C sequestration in the living biomass of trees by ca. 8 % in comparison with pre-industrial atmospheric concentrations of ozone. In a separate case study for temperate and boreal forests in northern and central Europe we calculated a similar decline of 10%. These values are of similar order of magnitude as the 7% reduction of total tree biomass at current ambient compared to pre-industrial 7 h mean ozone concentrations, determined in a meta-analysis (Wittig et al., 2009). From the case study in northern and central Europe it was concluded that the most important factor determining the changes in the forest living biomass C stock is the gap between growth and harvest rates.
- The concentration-based approach (AOT40) predicts substantially lower C reductions compared to the biologically more relevant flux-based approach (POD_y), especially in Northern and Continental Central Europe and when using RCA input data. By 2040, the AOT40 approach predicted a much smaller effect of ozone on C sequestration (a 2.1% reduction), mainly due to the decrease in mean hourly ozone concentrations to often below the threshold of 40 ppb (and hence not contributing to the calculated AOT40), whereas even concentrations lower than 30 ppb will still contribute to the calculated POD_y.
- As shown in previous studies, the spatial patterns of the concentration-based and flux-based method differ considerably across Europe. Whereas the concentration-based approach indicates a high risk of ozone impacts on forests in the Mediterranean areas, the flux-based approach indicates a high risk in Atlantic and Continental Central Europe, and also a higher risk in northern Europe. The concentration-based gradient from north to south Europe is greater than the flux-based gradient.
- A global modelling case-study using JULES (Joint UK Land Environment Simulator) and ozone stomatal flux-response relationships showed that the reduction in C stored in vegetation is 6.2% in 2000 compared to 1900, and is predicted to rise to 10.9% by 2040 due to a predicted rise in atmospheric ozone concentrations in the future emission scenario applied.
- In JULES, the plant damage due to ozone directly reduces plant photosynthesis, and thereby indirectly, leaf stomatal conductance. With elevated near surface ozone levels, the model simulates decreased plant productivity, and as less CO₂ is required for photosynthesis, reduced stomatal conductance. Therefore, the plant is able to preserve water supplies. Indeed, compared to 1900 elevated ozone levels predicted a rise in water runoff, 7.7% by 2000 and 12.6% by 2040.
- When the indirect effect of ozone on stomata via photosynthesis was switched off in JULES, stomatal conductance and river run off changes little over time (1900 – 2040). However, despite the difference in stomatal conductance response between simulations (i.e. indirect effect of ozone on stomata switched on or off), the differences in the response of the C cycle are rather modest.
- When applying a climate typical for dry years in Europe (e.g. 2003) then in the absence of a direct effect of ozone on stomatal conductance, the JULES model predicts that ozone acts to partially offset drought effects on vegetation by reducing stomatal conductance. However, the net impact of rising ozone concentrations is to further reduce plant productivity under drought.

8.3 Recommendations for policy development

- More stringent reductions of the emissions of precursors of ozone are required across the globe to further reduce both peak levels and background concentrations of ozone and hence reduce the growing threat from ozone pollution to C sequestration.
- It would be of benefit to better integrate policies and abatement measures aimed at reducing air pollution and climate change as both affect C sequestration in the future.
- Improved quantification of impacts of ozone within the context of climate change is urgently required to facilitate improved future predictions of the impacts of ozone on C storage in the living biomass of trees (national, regional, global).
- Stringent abatement policies aimed at short-lived climate forcers such as ozone provide an almost immediate benefit for their contribution to global warming, in contrast to long-lived climate forcers such as CO₂ and halocarbons, that stay in the atmosphere for a century or more.

8.4 Recommendations for further research

- There is an urgent need for more field-based, larger scale experiments where vegetation is exposed to multiple drivers of climate change for several years (at least one decade) to further investigate the overall impact of a combination of drivers of change on C sequestration in terrestrial ecosystems.
- Further development of the ozone flux-based method and establishment of robust flux-effect relationships for more species is required. Field-based ozone experiments should be conducted with more tree species, in particular in Mediterranean areas.
- Field-based ozone experiments should also include the assessment of ozone impacts on below-ground processes and soil C content.
- Further epidemiological studies on mature forest stands are required for the validation of existing and new ozone flux-effect relationships, often developed from exposure studies with young trees grown in pots.
- Experiments are needed on the interacting effects of climate change and ozone, including quantifying impacts of reduced soil moisture availability, rising temperature and incidences of heat stress, impacts of rising CO₂ concentrations and declining nitrogen deposition. Impacts of other drivers of change on existing flux-effect relationships should be investigated.
- Development of climate region-specific parameterisations for flux models to improve the accuracy of predictions.
- Existing flux models (e.g. DO₃SE) will have to be further developed to include more mechanistic approaches for the accurate prediction of combined effects of ozone, other pollutants and climate change on various plant physiological processes and hence C sequestration.
- There is an urgent need to further include ozone as a driver of change in global climate change modelling to quantify its impact (either directly or indirectly via impacts on vegetation)

on global warming. Such modelling should further investigate the mechanisms of interactions between ozone and other drivers of global warming.

- There is a need to quantify the economic impacts of ozone on forest growth in order to establish the economic consequences for the wood industry. In this light and for enhanced C storage in the living biomass in the future, ozone-sensitivity of tree species and varieties should be considered as a factor in future breeding and forest management programmes.

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Ozone pollution: Impacts on carbon sequestration in Europe

This report synthesises current knowledge on the effects of ground-level ozone on carbon sequestration in the living biomass of trees. Ground-level ozone is not only an important air pollutant, it is also the third most important greenhouse gas after carbon dioxide and methane. Both carbon dioxide and ozone are taken up through the thousands of microscopic pores on the surfaces of leaves. Whereas carbon dioxide is required for plant growth, ozone inhibits growth in sensitive species. Trees are an important sink for carbon dioxide and ozone, reducing their concentration in the atmosphere and hence the amount of global warming. However, the damaging impact of ozone on vegetation reduces the land carbon sink for these greenhouse gases, resulting in a positive feedback on global warming. For the first time, ozone effects in Europe have been quantified using the flux-based methodology that takes into account the modifying effect of climate, soil and plant factors on the amount of ozone taken up through leaf pores. Flux-based impacts of ozone on carbon sequestration in the living biomass of trees were estimated for the current (2000) and future climate (2040). This report also summarises the potential pathways via which ozone might affect carbon sequestration and how ozone impacts might interact in a future climate with elevated carbon dioxide, warming and increased drought episodes. Consequences for the global carbon and water cycle are described. Finally, recommendations are made for future policy aiming to protect vegetation from the adverse impacts of ozone on carbon sequestration.

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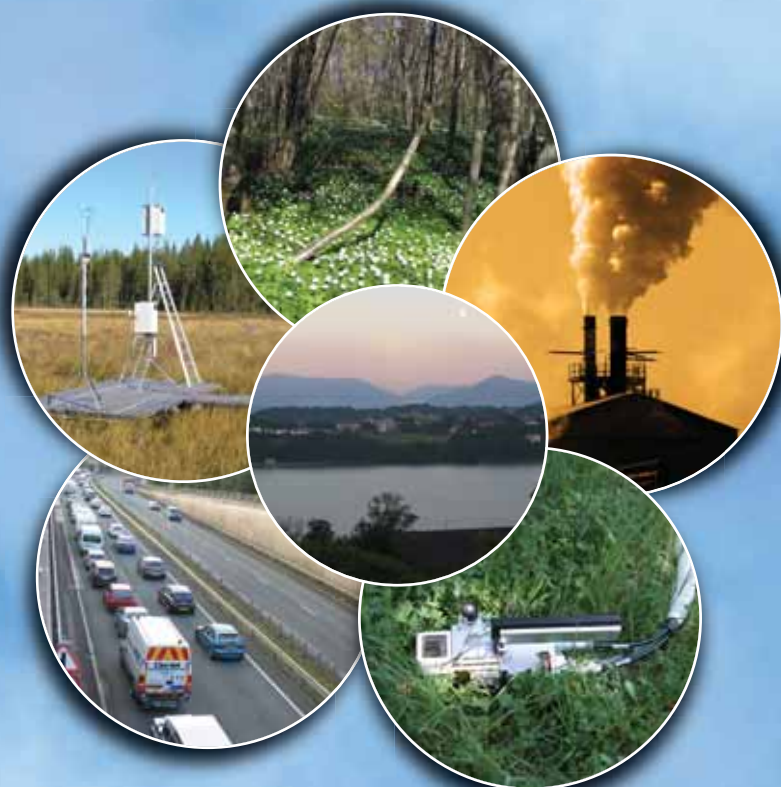
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